# Chapter Three

# Management and Control of Pathogens

## 3.1 Introduction

This chapter presents technical information supporting the Total Maximum Daily Load (TMDL) process for pathogens, specifically Step 5 – Allocations (U.S. EPA, 2001). The **allocations** step's objective (U.S. EPA, 2001) is to:

> **Using total assimilative capacity developed in the linkage component, develop recommendations for the allocation of loads among the various point and nonpoint sources, while accounting for uncertainties in the analyses (i.e., margin of safety) and, in some cases, a reserve for future loadings.**

The information provided will assist watershed managers in determining the capabilities of control technologies, i.e., disinfection, and best management practices (BMPs) for reducing microbial concentrations in point and nonpoint sources.

Following is the definition of point sources as presented in the Clean Water Act (CWA), Section 502 (14):

> The term "point source" means any discernible, confined and discrete conveyance, including but not limited to any pipe, ditch, channel, tunnel, conduit, well, discrete fissure, container, rolling stock, concentrated animal feeding operation, or vessel or other floating craft, from which pollutants are or may be discharged. This term does not include agricultural stormwater discharges and return flows from irrigated agriculture.

Wet weather flows (WWFs) regulated by the National Permit Discharge Elimination System (NPDES) program are considered point sources. These include combined sewer overflows (CSOs), stormwater associated with industrial activity, construction-related runoff, and discharges from municipal separate storm sewer systems (MS4s). MS4 stormwater types regulated through NPDES permits are described in Section 1.3.1.2. The CWA does not provide a detailed definition of nonpoint sources; these are defined by exclusion, i.e., anything not considered a point source in the CWA or EPA regulations. All nonpoint sources are caused by runoff of precipitation over or through the ground. Therefore, WWFs not covered through NPDES permits are nonpoint sources (U.S. EPA, 2003a).

As discussed in Chapter 1, there are detailed procedures and different approaches for determining recommended loads among the various point and nonpoint sources, while reserving a margin of safety and room for future loading increases. The TMDL definition is provided in Section 1.3.1.1. Development of a single waste load allocation (WLA) for all point sources of pathogens - publicly owned treatment works (POTW) or wastewater treatment plant (WWTP) effluents, and CSO, sanitary sewer overflow (SSO), and stormwater discharges - within one or more municipalities in a given urban watershed requires knowledge of treatment system capabilities and effective control strategies. Different approaches can be used to develop WLAs. One control strategy, a direct approach, is to calculate respective WLAs because treatment system capabilities and effective control strategies can be fully quantified. Another control strategy can be to sum up all the major sources of pathogen discharges. This approach provides the flexibility of adjusting the proportion of flow and loadings among any of the sources present, such as stormwater, CSO, SSO, and POTW or WWTP discharge locations, to maximize the treatment of sewage and load reductions. Point sources are generally discharged from a discrete point and are treated by control technologies and structural BMPs. However, there are discrete end-of-pipe or drainage channel conduit discharges that do not fit within the legal definition of a point source.[1]

Load allocations (LAs) consist of nonpoint sources and a natural background level of a given water body. WLAs and LAs pertaining to stormwater, CSO, and SSO occur intermittently as their origins are WWF events. Therefore, in establishing TMDLs, there needs to be a conversion of these intermittent loads into daily loads. Also, if there are known occurrences of untreated CSO and SSO discharges, these should be dealt with and accounted for independently.

LAs are established for nonpoint sources and, where necessary, may include implementation of BMPs and source reduction strategies. Discrete discharges and diffuse sources considered legally to be nonpoint sources can be managed using either control technologies or BMPs. Diffuse sources are generally managed through nonstructural BMPs. BMPs will be described in the latter part of this chapter (Section 3.3). In some cases, states have certain mandatory BMP requirements for specific land activities associated with large fecal indicator loads, such as confined animal operations or with flood control. Often, implementation of BMPs occurs through voluntary or incentive programs. Therefore, when establishing nonpoint source allocations within a TMDL, the documentation should include a reasonable assurance that the BMP(s) will be implemented and maintained and that the effectiveness of the BMP will be demonstrated. If pathogen loadings are to be reduced by a BMP, the TMDL strategy may require a long-term water quality monitoring program to demonstrate effectiveness of the BMP used.

---

[1] *The reader should be aware that "point source" is a legal term, as defined on page 3-1. It is also commonly used to describe all discrete discharges.*

The effectiveness of BMPs for controlling stressors in general, and pathogens, in particular, has not been fully established. There are few references with quantified pathogen removals. There is a difference between a treatment technology and a BMP (see Table 3-1).

**Table 3-1. Distinction between a Treatment Technology and a BMP for Pathogen Control**

| | Treatment Technology (Disinfection for Pathogens) | BMP |
|---|---|---|
| Source treated | discrete end-of-pipe or drainage channel conduit discharges | discrete end-of-pipe or drainage channel conduit discharges; diffuse sources |
| Effectiveness | known | uncertain; little data |
| Prediction of results | reasonably accurate | uncertain |
| Design | specific | specific |
| Improvement | to the level required | uncertain |
| Cost | known | known |

While the effectiveness and pollutant load reduction by a given BMP may be just an estimate, the effectiveness of a given technology is usually known and treatment results can be predicted with reasonable certainty. Although some structural BMPs can perform like treatment technologies, any misjudgement of treatment effectiveness will either reduce its usefulness and/or increase costs (Field, 1996).

The common practice for managing stormwater has been the use of structural and nonstructural BMPs. BMPs can achieve significant environmental improvements, such as reduction of flow volume and removal of suspended solids by sedimentation and filtration. BMPs achieve different degrees of removal of toxic substances and nutrients associated with the removed flow and solids. Removal of pathogens through the use of BMPs can also be associated with reduced flow and removed solids. Disinfection using treatment technologies is feasible for stormwater that can be collected and confined, but it is seldom implemented.

The following are three examples of collecting and treating stormwater or dry weather urban runoff:

1. The city of New Orleans, LA evaluated a prototype disinfection facility for stormwater using sodium hypochlorite in the late 1960s and early 1970s (Pavia and Powell, 1968); but they did not implement the practice permanently.

2. Santa Monica's urban runoff recycling facility (SMURRF, 2000) is treating dry weather runoff and some wet weather runoff since December 2000. (http://Epwm.Santa-Monica.Org/Epwm/Smurrf.html).
3. Moonlight Beach urban runoff treatment facility in the City of Encinitas, CA has been treating dry season urban runoff since September 2002 (Rasmus and Weldon, 2003).

## 3.2 Disinfection Technologies for Control of Pathogens

### *3.2.1 Introduction*

As long as satisfactory levels of suspended solids concentration and particle size are achieved upstream, disinfection technologies can achieve effective reduction of pathogen-contaminated concentrated sources such as:

1. POTW or WWTP effluent; sometimes referred to as secondary effluent
2. CSO, SSO, and stormwater discharges, all referred to as WWF because these discharges occur during wet weather events
3. Industrial wastewater discharges
4. Confined animal feeding operations (CAFOs)

While disinfection of WWTP effluent (or secondary effluent) and of industrial wastewater discharges is an established practice (U.S. EPA, 1986a), achieving disinfection of WWF is difficult. Because WWF is a significant contributor of microbial contamination to receiving waters, disinfection of WWF released as point sources is warranted.

As stated above, WWF point sources consist of CSO, SSO, and stormwater. Stormwater draining directly into a receiving water body, rather than through a sewerage system, also falls under the definition of WWF and may be considered to be either a point source or a nonpoint source. Human fecal contamination is the main concern for sanitary sewer systems. For stormwater systems, nonhuman-origin (other warm-blooded organisms) and human-origin fecal coliform microbial contamination from unauthorized sanitary sewage cross-connections are the concerns. In combined sewer and storm drainage systems, fecal contamination of both human- and non-human origin are of concern.

Since issuance of the National CSO Control Policy (U.S. EPA, 1994), which requires disinfection of CSO after primary clarification, the CSO became the most frequently disinfected component of WWF. Most WWF disinfection studies, with conventional and alternative technologies, have been conducted on CSO (U.S. EPA, 2002a). However, all components of WWF, such as SSO and stormwater, carry significant loads of fecal and pathogen contamination that would be reduced by disinfection.

Numerous factors need to be considered in discussing WWF disinfection:

1. Disinfection effectiveness as demonstrated by the pathogen reduction levels
2. The need for a high-rate disinfection process
3. The need for suspended solids removal prior to disinfection
4. A description of individual disinfection technologies in the diminishing order of their commercial availability for WWF treatment and their relative costs
5. A description of disinfection studies and implementation examples

### *3.2.2 WWF Disinfection Effectiveness*

Disinfection effectiveness is conventionally judged by the reduction of microorganism densities, generally a bacteriological indicator. Disinfection technologies have been tested using a variety of bacterial and viral indicators and selected individual pathogenic organisms as well. Where available, this information is presented in the subsequent sections on individual disinfection technologies. Different indicators may respond very differently to the disinfection process. A study by the Massachusetts Water Resources Authority, Boston, MA compared *Enterococcus* to fecal coliform data in secondary treated effluent and in effluent from CSO facilities. The investigators found significant differences between how the indicators respond to treatment. Satisfactory reduction of fecal coliform was achieved with chlorination, but the reduction of *Enterococcus* was unsatisfactory (Rex, 2000).

Development of bacteriological indicators was necessitated by the fact that it is both impractical and expensive to isolate and measure specific pathogenic organisms in water. Use of the various indicators is discussed in Chapter 1 and summarized here. A group of enteric bacteria known as coliform are plentiful in human wastes and easy to measure. Therefore, bacteria of the total coliform group became the generally accepted indicator for fecal pollution, even though this group includes different genera that do not all originate from fecal wastes (e.g., *Citrobacter*, *Klebsiella*, and *Enterobacter*). An improvement over the total coliform indicator is the more selective fecal coliform indicator, since fecal contamination of human origin is known to cause diseases in humans. Fecal coliform selects primarily for *Klebsiella* and *Escherichia coli* (*E. coli*) bacteria. *E. coli* is the bacterium of interest because it is a consistent inhabitant of the intestinal tract of humans and other warm-blooded animals. However, the fecal coliform test is still not fully specific to enteric bacteria and human-enteric bacteria in particular (O'Shea and Field, 1992).

As discussed in Chapter 1, stormwater runoff can contain high densities of the non-human indicator bacteria, and epidemiological studies of recreational waters receiving stormwater runoff have found little correlation between fecal coliform indicator densities and swimming-related illnesses (U.S. EPA, 1984; Calderon *et al.*, 1991). In 1986, U.S. EPA recommended that states begin the transition process to the *E. coli* and *enterococci* indicators (U.S. EPA, 1986b). However, many states still retain the total and fecal coliform criteria. The most widely used bacteriological criterion in the U.S. is a maximum of 200 fecal coliform/100 mL in waters designated for swimming (Field, 1990). Because the fecal coliform indicator is the most widely used, disinfection effectiveness is often reported as reduction of this indicator. Untreated WWF may contain densities of $1\times10^5$ to $1\times10^7$ fecal coliform/100 mL. Achieving hundreds ($10^2$) of fecal coliform/100 mL in treated WWF with the use of a given disinfection technology would indicate a very successful treatment. Achieving thousands ($10^3$) of fecal coliform/100 mL in treated WWF with the use of a given disinfection technology may still indicate an adequate treatment if there will be a significant dilution upon discharge of the treated effluent.

### *3.2.3 Requirement for a High-Rate Disinfection Process*

Experience has shown that the long contact time required for conventional wastewater treatment is extremely costly for the treatment of WWFs due to their relatively high flow rates and intermittently occuring volumes. However, WWF disinfection can be achieved at shorter contact times. (U.S. EPA, 1979a; U.S. EPA, 1979b; Stinson *et al.*, 1998). This approach has been termed "high-rate disinfection." There is currently no clear definition as to what constitutes high-rate disinfection other than achieving the required bacterial reductions at detention times significantly less than 30 minutes, the standard contact time (U.S. EPA, 1993).

High-rate disinfection is accomplished by: (1) increased mixing intensity, (2) use of higher concentrations of disinfectant, (3) use of chemicals or irradiation with higher oxidizing rates or microorganism-kill potential, or (4) combinations of these (Field, 1990). The use of increased mixing with any disinfection technology provides better dispersion of the disinfectant and forces disinfectant contact with a greater number of microorganisms per unit time. The increased rate of collisions decreases the required contact time enabling high-rate disinfection (Glover, 1973). An effective disinfection process will have to provide the desired microbial deactivation very rapidly under the specific WWF conditions and carry an insignificant amount of disinfectant residual into the receiving water.

### *3.2.4 Requirement for Suspended Solids Removal*

Effective use of any disinfection technology on WWF requires use of a treatment train, where its initial segment removes excess suspended solids and its final segment is the disinfection process. WWF disinfection requires some form of filtration or clarification/sedimentation prior to introduction of disinfecting chemicals (U.S. EPA, 1973). High levels of particulate matter in WWF can provide a "shielding effect" in which particles present in the medium protect the microbes either from disinfecting agent. (Sakamoto and Cairns, 1997).

Microbial aggregation and particle association are two phenomena that protect microbes and, thus, are major causes of decreased disinfection efficiency. Microorganisms have a tendency to clump together to form aggregates. While the organisms living on the outer layer of the aggregate can be easily disinfected, the microbes living inside are only partially (if at all) penetrated by the disinfectant or by UV light (Katzenelson *et al.*, 1976). Particle association can be represented by attachment of the microorganisms to the particle's surface and by microbial occlusion within the particle. Microbes attached to the particle's surface are usually properly disinfected but microbes occluded or hidden within the particles may not be disinfected at all.

Studies have shown that pretreatment processes (e.g., filtration, sedimentation) can significantly reduce the effects of both aggregation and occlusion on disinfection efficiency. Johnson *et al.* (1983), for instance, tested both filtered and unfiltered secondary wastewater effluents that were subjected to UV disinfection in side-by-side UV reactors. The filtered effluent showed significantly better disinfection than the unfiltered medium. The study concluded that microbial protection by large particle occlusion is the major reason for increased disinfection efficiency after filtration. Therefore, particle count and size distribution are important indicators of the influent quality and its need for pretreatment. Particularly sensitive to suspended solids content is UV disinfection, which is significantly more effective at suspended solids contents of less than 150 ppm (U.S. EPA, 2002b). UV disinfection tested on CSO and SSO after compressed media filtration (Fuzzy Filter) showed improved performance (U.S. EPA, 2002c). In case of chemical disinfection, the lower suspended solids content in the treated effluent, the less chemical addition and shorter contact times are needed for effective disinfection.

### *3.2.5 WWF Disinfection Technologies*

Alternatives to chlorination disinfection technologies, for example UV light irradiation, chlorine dioxide ($ClO_2$), and ozonation ($O_3$), generate significantly less toxic byproducts and residuals when compared to chlorine ($Cl_2$). However, only chlorination/dechlorination, as opposed to alternative technologies, is currently used for WWF disinfection. There have been several pilot studies on WWF with alternative technologies. The Water Environment Research Foundation (WERF) were sponsoring a study that evaluates the risks and benefits associated with various CSO disinfection technologies. A report on its results will be published by 2004. Disinfection technologies are discussed below in diminishing order of their commercial availability for WWF treatment.

#### 3.2.5.1 Chlorination and Dechlorination

Disinfection by $Cl_2$ has proven to be effective, and has been used for wastewater disinfection in the U.S. since 1855 (White, 1999). Chlorine or its derivatives are the most commonly applied disinfectants in the U.S. (SAIC, 1998). Chlorine is readily available in several forms, inexpensive, and effective against bacteria, though not fully effective against viruses. Chlorine is ineffective in killing protozoa. The easiest way to increase $Cl_2$ effectiveness is to increase the $Cl_2$ dosage within the system. This, however, results in the additional generation of toxic, carcinogenic, and/or mutagenic byproducts, as well as a high residual concentration of $Cl_2$ in the receiving waters. In the last 20 years, disinfection by chlorination has come under scrutiny. Research studies, particularly for drinking water, have cited health risks with regard to $Cl_2$ and its byproducts. Excess of free $Cl_2$ can cause chlorinated hydrocarbon formation, i.e., chloroform and trihalomethanes (THMs), which are suspected carcinogens. Chlorine residuals discharged to natural waters may be harmful to aquatic life.

Disinfecting high volumes of WWF requires large quantities of $Cl_2$. Because of the high risk of gas leaks when transporting gaseous $Cl_2$, use of liquid $Cl_2$ in the form of calcium hypochlorite and sodium hypochlorite is preferred but more expensive. Liquid $Cl_2$, as sodium or calcium hypochlorite, is easier to handle, safe to store in onsite tanks, and immediately available for use. The effectiveness of liquid versus gaseous $Cl_2$ for disinfection of WWF has been investigated. In general, the studies confirmed that liquid $Cl_2$ is a better disinfectant for WWF, and WWF facilities are encouraged to changeover from gaseous to liquid $Cl_2$. When necessary, excess of free $Cl_2$ can be removed by using either gaseous sulfur dioxide or sodium bisulfite solution. This will eliminate further byproduct formation, but will neither eliminate nor reduce the already-formed harmful byproducts. Dechlorination also means the addition of another process, which raises the cost of disinfection. On the average, dechlorination will add about 30% to the total cost of disinfection. After dechlorination, there is an analytical challenge in measuring the required low residual level of $Cl_2$ and the associated monitoring of $Cl_2$ levels in receiving waters.

The chlorination/dechlorination pilot study at the 26$^{th}$ Ward WWTP testing facility in New York City, NY demonstrated that hypochlorite disinfection was a cost-effective technology for the upgraded Spring Creek facility because of the existing tanks at this facility. Dechlorination will be added at a later date. Improvements will be made to increase disinfectant flash mixing and to automate hypochlorite feed and control of the residual chlorine (U.S. EPA, 2002b). The study is described in greater detail in Section 3.2.6.1.

Chlorination/dechlorination of CSO was tested on over 40 wet-weather events at a full-scale Advanced Demonstration Facility (ADF) in Columbus, Georgia. This study is summarized under the 3.2.6.3 subsection of this Chapter. Detailed performance results and relative costs are presented in a report (Columbus Water Works, 2001).

#### 3.2.5.2 Ultraviolet Light Irradiation

Since the early 1900s, UV light irradiation from mercury arcs has been recognized as an efficient disinfecting agent. At the germicidal wavelengths, within the range of 200 to 320 nanometers (nm) wavelength, UV light disinfects water by altering the genetic material in microbial cells, preventing reproduction. Peak effectiveness occurs near 253.7 nm, the wavelength of emission from a mercury arc lamp. UV irradiation has become an acceptable alternative to chlorination for wastewaters undergoing a secondary or tertiary treatment. Until recently, it has not been used for low-quality effluents such as WWF as an alternative to chlorination.

Certain parameters determine the UV dose required for effective disinfection. Understanding these parameters and their variability in WWF is very important for proper disinfection system design (Ashok *et al.* 1997). High variability in WWF flow rates influences UV disinfection effectiveness, because flow rate is a principal determinant of the dosage of UV light necessary for effective disinfection (Wojtenko *et al.,* 2001a). This is generally true for all WWF disinfectants, but UV disinfection is more affected by wastewater quality than chemical disinfection technologies. High levels of suspended solids containing particles larger than 2 microns and minerals present in WWF also reduce UV light effectiveness. During disinfection, the negatively charged quartz sleeves surrounding the UV lamps foul by picking up positive ions (e.g., Ca, Mg, and Fe) from the water. Fouling materials decrease transmittance of UV light and thus its disinfection capability (Oliver and Gosgrove, 1975). Use of an in-place cleaning system can remove fouling materials from the quartz sleeves.

Using UV irradiation for disinfection eliminates many problems arising from chlorination, such as the need for chemicals and their associated transportation, handling, and storage, as well as the need for expensive dechlorination facilities. Eliminating large contact tanks and facility buildings significantly lowers capital and operating costs. UV light irradiation affects a wide range of microorganisms and does not generate known harmful secondary chemical byproducts (e.g., THMs). Based on investigations, UV light irradiation for CSO disinfection shows promise as an effective and safe alternative to chlorination. To inactivate the target microorganisms efficiently, UV light must penetrate the water. Therefore, the water to be disinfected must be as clear as possible.

High levels of particulate matter in WWF absorb a large amount of energy, significantly decreasing the amount of UV light available for disinfection. UV light can disinfect free-living microorganisms very effectively with a low dose of irradiation, but microbes are often adsorbed to surfaces of particles (e.g., soil, sediment) or embedded within solid materials (e.g., fecal material). Solid particles shield the microbes from the disinfecting agent. Adsorption of the microorganisms to inorganic surfaces does not affect disinfection efficiency as significantly as adsorption to organic matter. The presence of a surface like clay does not inhibit UV disinfection because it tends to scatter UV light rather than absorb it.

UV light irradiation is a physical procedure that does not alter the smell or chemical composition of water. UV disinfection for WWF requires some level of physical pretreatment (with or without chemicals) to make UV light more effective for WWF (Field, 1996). Pilot studies have shown that filtration prior to UV disinfection can minimize the effects of particle occlusion/association (Johnson *et al.*, 1983).

In a 1996 pilot study of high-rate CSO treatment technologies in the Metropolitan Toronto Area, Canada, UV disinfection was used to achieve an *E. coli* count of 200 cfu/100 mL in a CSO effluent treated by a vortex separator, marketed as the Storm King. UV collimated beam tests were undertaken on only two samples, and in both cases the vortex separator was operated at a surface load of 10 m/h, with a cationic polymer dosage of 8 mg/L. The residual total suspended solids (TSS) in vortex effluent samples averaged 42 mg/L and the interim target fecal coliform count had been achieved at a UV dosage of 30 $mWs/cm^2$, which was considered to be a feasible dosage for full-scale application. The cationic polymer coagulant was used to improve the solid/liquid separation efficiency and, thus, facilitate UV disinfection (Marsalek *et al.*, 1996).

UV testing on CSOs in the ADF study in Columbus, GA, was also done in a treatment train arrangement. UV was tested after both vortex and compressed media filtration and its performance was better on the filtered CSO than on the unfiltered CSO. This study is summarized under the 3.2.6.3 subsection of this chapter. Detailed performance results and relative costs are presented in a report (Columbus Water Works, 2001). UV testing after the use of compressed media filtration (Fuzzy Filter) was done on SSO-type wastewater at the Rockland County, NY sewer district testing facility. This study is summarized under the 3.2.6.3 subsection of this chapter (U.S. EPA, 2002c).

Investigations of UV light irradiation for CSO disinfection have shown this technology to have the potential to be an effective and safe alternative to chlorination, assuming the adequate removal of suspended solids prior to UV application. A CSO disinfection pilot study conducted at the 26$^{th}$ Ward WWTP testing facility in New York City that evaluated and compared UV light, $O_3$, $ClO_2$, and chlorination/dechlorination disinfection units showed that the UV light unit was the simplest unit to operate. This study is summarized under the 3.2.6.3 subsection of this chapter (U.S. EPA, 2002b).

It is evident from studies and implementation examples described under section 2.6.3, UV technology has been gaining acceptance for treatment of CSO.

#### 3.2.5.3 Chlorine Dioxide

The use of $ClO_2$ for WWF disinfection has also been investigated as an alternative to chlorination. The lack of any significant reactions of $ClO_2$ with water is the main reason for its biocidal effectiveness over a wide pH range. Chlorine dioxide was found to provide excellent disinfection at a fraction of the $Cl_2$ dosage, making it cost effective and relatively safe. In addition to its high effectiveness over a wide pH range, the low reactivity of $ClO_2$ with ammonia and reduced formation of halogenated organic compounds are its major advantages over $Cl_2$. However, the presence of organic and inorganic impurities in water is a limitation of $ClO_2$ disinfection. The impurities create a large oxidation demand for $ClO_2$. These reactions take place together with disinfection (Katz *et al.*, 1994). In such a system, the effectiveness of the disinfecting agent is greatly reduced. Effective treatment of the wastewater by filtration and/or sedimentation is a precursor for successful $ClO_2$ disinfection (Stinson *et al.*, 1999). This is of great importance for CSO applications.

Chlorine dioxide is a very strong and effective wastewater disinfectant. It is not a chlorinating agent and does not lead directly to the formation of organochlorine compounds (Dernat and Pouillot, 1992). The major advantages of $ClO_2$ are: its disinfection effectiveness for $Cl_2$-resistant pathogens (e.g., viruses and protozoa) within a wider pH range, its high solubility in water, the production of stable and measurable residue, and no reactivity with ammonia to produce chloramines. Due to these advantages, $ClO_2$ was found to be an attractive candidate for WWF applications. Because $ClO_2$ is a more powerful disinfectant than $Cl_2$, lower levels of $ClO_2$ can be used resulting in lower levels of toxic byproducts to get the same level of inactivation.

For several decades, researchers have compared the respective disinfection efficiencies of $ClO_2$ and $Cl_2$. In potable water as well as in wastewater treatment applications, a number of researchers have found a significantly lower $ClO_2$ demand compared to that of $Cl_2$. In studies where equivalent amounts of each of the disinfectants were added to water with various levels of contamination, after 30 minutes of contact, $Cl_2$ was found to be largely consumed while $ClO_2$ remained mostly unreacted. This result indicates that $ClO_2$ reacts with fewer compounds than $Cl_2$. Due to the limited reactions of $ClO_2$ with organic compounds in water, more of the disinfectant remained available as a biocidal agent. Chlorine dioxide was found to be a stronger disinfectant than $Cl_2$ at shorter contact times and, in addition, was found effective against a greater number of different microorganisms (Moffa, 1975). Chlorine dioxide was also found to be a better disinfectant of bacteria and more effective than $Cl_2$ against viruses and protozoa (Aieta *et al.*, 1980).

The possibility of using a combination of $ClO_2$ and $Cl_2$ was investigated for municipal wastewater treatment by Katz *et al.* (1994). After adding both agents in equal amounts, improved disinfection efficiency was observed with all doses, and production of the byproducts, such as chlorite ion ($ClO_2^-$) and THMs, was greatly reduced. Chlorine dioxide used in combination with $Cl_2$ also resulted in a lower residual $Cl_2$ concentration. A bench-scale study was conducted by the U.S. EPA on high-rate disinfection using $Cl_2$ and $ClO_2$ and its findings were verified by two full-scale prototype treatment facilities for CSOs (Moffa, 1975). The concentration of residual $ClO_2$, increased while the concentration of toxic $ClO_2^-$ decreased. This was explained by Katz *et al.* (1994) as being the result of an oxidation reaction between $Cl_2$ and $ClO_2^-$ to produce $ClO_2$. When the combination of $ClO_2$ and $Cl_2$ is used, $ClO_2$ competes with $Cl_2$ for the oxidation of organic precursors to THM and chloroorganic compounds. Chlorine reduced the concentration of $ClO_2^-$ by oxidizing it back to $ClO_2$. In this case, $Cl_2$, the cheaper disinfectant, increased the concentration of $ClO_2$, the more expensive disinfectant, thus lowering the cost of the disinfection process.

Despite the numerous advantages of $ClO_2$ disinfection, the necessity for $ClO_2$ generation onsite due to its instability is a major disadvantage. The most commonly used $ClO_2$ generation method is the reaction of $NaClO_2$ with acid (White, 1999). There are safety considerations associated with $ClO_2$ generation: instability of $ClO_2$ as a gas, storage and transport of its precursors (e.g., gaseous $Cl_{2\,,}$ sodium chlorite $NaClO_2$) on site, and proper operation of the equipment. There is a serious problem with a delivery of gaseous $Cl_2$ as it is prohibited to be transported through most densely populated areas. There is a new process of $ClO_2$ generation that uses $NaClO_2$ in the presence of UV light (Stinson *et al.*, 1998). In this process the transport and handling of gaseous $Cl_2$ is totally eliminated but this process is still under development and is not commercially available. Other disadvantages of $ClO_2$ disinfection include lack of data available for full-scale application to WWF and the potential explosion hazard under certain conditions.

The potential advantages of using $ClO_2$ as a disinfectant greatly outweigh the possible disadvantages and inconvenience of onsite generation. When produced, handled, and used properly, $ClO_2$ is an effective and powerful disinfectant. The sequential addition of $Cl_2$ with

$ClO_2$ greatly enhances the disinfection process and is cost effective. Chlorine dioxide appears to have potential for becoming an effective $Cl_2$ alternative for WWF disinfection. Further investigations, however, are recommended to determine its effectiveness in a full-scale WWF application (U.S. EPA, 2002b).

Chlorine dioxide performed better than chlorination/dechlorination in the Columbus ADF study (Columbus Water Works, 2001) and in the New York City study (U.S. EPA, 2002b). Of particular interest was the second phase of the New York City study where a new process of $ClO_2$ generation using UV light, which avoids the need for gaseous $Cl_2$, was used as the source of $ClO_2$. While $ClO_2$ was superior in effectiveness and similar in cost to chlorination/dechlorination, the UV generation technology for $ClO_2$ needs further development. Currently, $Cl_2$ gas cannot be transported within New York City. Thus, because an effective $Cl_2$-gas-free process of $ClO_2$ generation has not been proven to be reliable, disinfection with $ClO_2$ cannot be considered for use within New York City, or any other metropolitan area, at this time. The Columbus ADF study is summarized under the 3.2.6.3 subsection of this chapter. Detailed performance results and relative costs are presented in a report (Columbus Water Works, 2001). The New York City study is summarized under the 3.2.6.3 subsection of this chapter. (U.S. EPA, 2002b).

#### 3.2.5.4 Ozonation

Ozone's ability to inactivate microorganisms was already well known as early as 1886 (White, 1999). It is the strongest and fastest-acting oxidant of all the classical disinfecting agents used for water sanitation today. Ozone inactivates a wider range of microorganisms than $Cl_2$, has a relatively high disinfection-kill power, releases limited byproducts, has the ability to increase dissolved $O_2$ concentration, is non-reactive with ammonium, and has an excellent ability for removing undesirable odor and color. In addition to being a strong disinfectant, $O_3$ reacts with organic impurities (e.g., saturated hydrocarbons, amines, and aromatic compounds) destroying them in the process and forming such byproducts as acids, aldehydes, bromates, ketones, and peroxides. Studies evaluating ozonation byproducts are limited, and further investigation in this area is necessary.

Because $O_3$ is a very strong oxidant, it has the potential for being effective for low-quality wastewater and WWF disinfection. Organic and inorganic impurities, chemical oxygen demand (COD), pH, temperature, and suspended solids in waters have a significant impact on $O_3$ disinfection efficiency. The presence of water impurities is a major limiting factor of ozonation for CSO applications. As a strong oxidant, $O_3$ will react with many organic (e.g., aromatic and aliphatic compounds, pesticides, humic acids) and inorganic (e.g., sulfide, nitrogen, iron, manganese, cyanide) compounds producing reaction byproducts (U.S. EPA, 1986a). Reactions with impurities consume $O_3$, which is then no longer available as a disinfecting agent. As a result, wastewater with high levels of impurities requires a high dosage of $O_3$ and, thus, an increased $O_3$ demand, for disinfection to be successful. Although $O_3$ is a strong oxidant and a powerful disinfectant, its application for WWF disinfection has been very limited. As indicated by White (1999), effective ozonation requires relatively good water quality; thus, filtration is recommended before the ozonation process.

Similar to every other disinfection process, ozonation is most effective for free-floating organisms. The presence of particles in water makes ozonation challenging. In addition to particle occlusion, microbial aggregation was also found to be a major factor negatively affecting ozonation. The rates of inactivation of aggregates were found to be much slower when compared to free organisms.

The equipment and operating costs associated with ozonation are relatively high. Due to its high instability, $O_3$ must be produced onsite and used within a short period of time. Skilled operators and constant attention are required. The necessity for onsite generation makes its application to the intermittent nature of WWF difficult.

In general, the ozonation process, if properly run, can be successful for disinfection of various water qualities (wastewater and drinking water). The CSO disinfection pilot study in New York City showed that there are some safety issues with $O_3$ generation and use, such as collection of off-gas and destruction of $O_3$, use of water-tight and gas-tight contactors, proper monitoring of the ventilation system, and use of corrosion-resistant construction materials (Stinson *et al.*, 1998; U.S. EPA, 2002b). Ozone instability is also a major factor contributing to the high cost of this technology. There are currently no WWF facilities using this technology in the U.S.

In the New York City study, the capital costs of $O_3$ generation were found to be the highest of all technologies that had been investigated concurrently. Costs of ozone disinfection were found to be dependent on the cost of electricity as well as the source of oxygen used as a feed (air vs. pure $O_2$). This study is summarized under the 3.2.6.3 subsection of this chapter. (U.S. EPA, 2002b).

### *3.2.6 Description of Disinfection Studies and Implementation Examples*

#### 3.2.6.1 Disinfection Pilot Study at the 26$^{th}$ Ward WWTP Testing Facility in New York City

This pilot study demonstrated alternatives to hypochlorite disinfection for application to the Spring Creek CSO storage facility and potentially to other CSO facilities. The pilot testing was divided into two phases. Phase I evaluated treatment performance of five technologies: UV, $O_3$, $ClO_2$, chlorination/dechlorination, and electron beam irradiation (E-Beam). Based on the results from Phase I, Phase II provided additional evaluation of technologies that had shown potential for CSO applications. These were UV, $ClO_2$, and chlorination/dechlorination.

***Major findings***

- With the exception of E-beam, the tested technologies achieved targeted bacterial reductions of 3 to 4 logs.

- Chlorination/dechlorination, $ClO_2$, and $O_3$ provided targeted levels of disinfection over the full range of wastewater quality tested.
- Chlorine dioxide was superior in effectiveness and similar in cost to chlorination/dechlorination. The new technology for $ClO_2$ generation that does not require use of chlorine gas needs further development.
- The upgraded Spring Creek facility will continue to use sodium hypochlorite for disinfection, with provisions to add dechlorination at a later date. Improvements will be made to increase disinfectant flash mixing and to automate hypochlorite feed and residual control.

***Wastewater quality***

Five disinfection technologies, UV, $ClO_2$, $Cl_2$, $O_3$, and E-Beam, were tested for their effectiveness in reducing bacteria levels in water representative of the CSO at the Spring Creek storage facility. Tests were conducted during wet and dry events. To achieve a four-log reduction of fecal coliform and a fecal coliform effluent concentration less than 1,000 colony forming units/100 mL (cfu/100 mL) required doses for UV, $O_3$, $ClO_2$, and $Cl_2$ of 60-80 mWs/cm$^2$, 24 mg/L, 8-10 mg/L, and 20-28 mg/L, respectively. The range of disinfectant doses for each technology reflects the variation in performance between Phase I (December through March) and Phase II (August through November). The variation in wastewater temperature between Phase I (mean of 11.6 $^oC$) and Phase II (mean of 20.9 $^oC$) had a significant impact on the performance of $Cl_2$ disinfection. The colder winter temperatures impede the formation of monochloramine, which has approximately 25 times less germicidal efficiency than free $Cl_2$.

***Treatment Performance***

Four bacteria indicators were used as a measure of the effectiveness of each of the disinfection technologies; namely total coliform, fecal coliform, *E. coli*, and *Enterococcus*. Kills of each of the indicators, in terms of log reduction and concentration, were related to dose for each of the disinfection technologies. Chlorination/dechlorination, $ClO_2$, and $O_3$ at the doses tested were able to provide the disinfection levels of the four-log reduction over the full range of wastewater quality tested. UV disinfection effectiveness tended to drop off at higher TSS concentrations (e.g., TSS greater than approximately 150 mg/L). This was attributed to lower effective penetration of UV due to harboring of bacteria in solids.

Fecal coliform and *E. coli* exhibited similar dose-response relationships. However, total coliform and *Enterococcus* generally required higher doses to achieve the same level of inactivation as that for fecal coliform and *E. coli.* This was observed in all technologies except for the E-beam, where the inactivation results were inconclusive.

The UV and $ClO_2$ technologies provided nearly complete reductions of bacteriophage, a bacterial virus and microbial indicator. However, the viral inactivation data for the $ClO_2$ system was limited to only two of the four runs due to operational problems. Of the valid data considered, the effluent concentrations of bacteriophage ranged from non-detect to 60 cfu/mL. Low influent concentrations of the seeded phage limited the maximum log reduction that could be observed.  The log reduction of bacteriophage ranged from 1.9 to 6.0.  Because of the low concentrations of naturally occurring enteroviruses in the pilot influent, the UV disinfection could not be evaluated satisfactorily on the basis of the tissue culture infectivity assays, discussed in Chapter 2.  However, based upon the reductions of the marginal concentrations found and upon the bacteriophage results, these technologies would inactivate most natural enteroviruses found in wastewater at concentrations on the order of $10^6$ cfu/mL.

UV disinfection achieved 4-log bacteria reduction but at extremely high dosage levels owing to the impediments of poor water quality.  UV effectiveness tended to be reduced by high TSS concentrations (e.g., greater than 150 mg/L).  Additionally, UV effectiveness tended not to increase at doses greater than 75 mWs/cm$^2$, a phenomena known as “tailing-off.”

Ozone disinfection can be accomplished only at high $O_3$ dosage levels. However, the $O_3$ pilot unit did not include a contactor design appropriate for the wastewater conditions tested. Thus, the required $O_3$ dosages may have been less if a more applicable $O_3$ dissolution/contactor system were provided.  An $O_3$ disinfection system would require contact chambers other than the tankage that presently exists at Spring Creek.

Chlorine disinfection included dechlorination to eliminate residual $Cl_2$.  Chlorination as well as dechlorination can be accomplished using the existing tanks at the Spring Creek Advanced Wastewater Pollution Control Plant (AWPCP).  High-rate mixing can be added to the head end of the tanks.  Chlorine dioxide disinfection can be accomplished at doses on the order of 30% of the required $Cl_2$ dose.

Chlorination/dechlorination and $ClO_2$ were determined to be the most cost effective technologies for application to Spring Creek.  However, neither of the $ClO_2$ generation methods tested are currently feasible for use within New York City.  The $Cl_2$ gas/solid sodium chlorite generation method is not feasible because of its use of $Cl_2$ gas, and the UV/sodium chlorite generation method is not feasible because of its developmental status as a prototype. The capital costs for UV and $O_3$ were significantly more expensive than chlorination/dechlorination or $ClO_2$. For other CSO facilities that do not have existing tanks for contact time, UV could be economically attractive.

In the case of $ClO_2$, there is no significant increase in disinfection performance beyond a contact time of three minutes. This is in contrast to the chlorination results, which show a greater dependence on contact time and required five minutes for comparable kills.  The difference is attributed to $ClO_2$’s greater bactericidal properties and solids penetration characteristics than those of chlorination.  The results of this study confirm the optimum contact times for $ClO_2$ and chlorination/dechlorination  of three and five minutes, respectively, originally determined in the Syracuse and Rochester studies (U.S. EPA, 1979a and 1979b).  Chlorination/dechlorination and $ClO_2$ were determined to be the most cost-effective technologies for application at this facility. Further development of the UV-chlorite $ClO_2$ generator is required before reliable costs for this technology can be developed.

### *Disinfection Residuals and Toxicity*

Selected disinfection effluent residuals and byproducts, namely $ClO_2$, chlorate, chlorite, total residual chlorine (TRC), volatile and semivolatile organics, haloacetic acids, were monitored to relate these residuals to disinfectant dose. UV disinfection had the distinct advantage of producing no byproducts. This is in contrast to $Cl_2$ and $ClO_2$, which produced increased levels of TRC, chlorate, chlorite and haloacetic acids in the effluent. The slightly increased haloacetic acid concentrations were considered insignificant. The increased TRC, chlorate and chlorite concentrations were directly related to increased $Cl_2$ and $ClO_2$ dose.

No additional toxicity was observed in the UV effluent as compared to the UV pilot influent. In contrast, there were occurrences where the $ClO_2$ effluent was considerably more toxic than the pilot influent. An attempt was made to correlate this toxicity with the specific disinfection byproducts, in particular TRC, chlorate and chlorite, but no correlation could be made. It is likely that the increased effluent toxicity is directly related to influent toxicity (i.e., influent water quality) or a synergistic effect of the disinfectant residuals, which could not be measured. Although the concentrations of TRC, chlorate and chlorite did not cause concern about effluent toxicity, this relationship should be revisited when establishing $ClO_2$ dose for specific sites.

Effluent TRC was generally below 0.1 mg/L following dechlorination as compared to a receiving water quality standard of 0.0075 mg/L. This TRC value of dechlorinated effluent reflects the practical quantitation limit of the process instrumentation used. Lower TRC values could not be quantified. Often, the dechlorinated effluent TRC instrumentation displayed a negative value indicating the presence of excess bisulfite. Residual $Cl_2$ was also monitored in the $ClO_2$ effluent. However, these TRC values include all oxidizing species of $Cl_2$ and the possible presence of free and combined $Cl_2$ could not be differentiated from $ClO_2$, $ClO_2^-$ and $ClO_3^-$.

### *Chlorine Dioxide Generation*

The method of generating $ClO_2$ must be considered when selecting the appropriate disinfection process. The $Cl_2$ gas/solid sodium chlorite generation method was tested during the Phase I and Phase II pilot studies. Although this pilot unit was reliable, the use of $Cl_2$ gas (either with $Cl_2$ cylinders or with on-site $Cl_2$ gas generation) in this process may limit its application in residential and urban areas, including New York City. The UV-sodium chlorite solution generation method was also tested during the Phase II pilot study. This method had the distinct advantage of not using or generating chlorine gas in the generation process. However, this technology is currently in the prototype stage of development and would need to be developed as a full-scale unit to be considered further. The UV-chlorite generator from the UVD, Inc., was a prototype unit.

*Cost Comparison*

During the Phase I pilot study, conceptual level cost projections were prepared for each disinfection technology for comparison purposes, with the goal of recommending a technology for implementation at the Spring Creek storage facility. The Phase II pilot study results served to verify the Phase I result; as such, the assumptions and approach used for the original cost comparison were applicable. Costs for each disinfection technology were prepared on a common flow basis and were prepared for a range of flow rates experienced at Spring Creek. See Table 3-2. This approach shows the sensitivity of cost to flow rate, and allows independent comparison of technology costs at similar flow rates. Equipment capital costs were developed for peak design flow conditions of 1,250 cubic feet per second (cfs) (800 million gallons per day (mgd)), 2,500 cfs (1,600 mgd), and 5,000 cfs (3,200 mgd) for a duration of 4 hours. (U.S. EPA, 2002b).

### 3.2.6.2 Continuous Deflection Separation, Fuzzy Filter and UV Treatment of SSO-Type Wastewaters: Pilot Study Results

This study evaluated three technologies for treatment of SSO and CSO overflows. These were the Continuous Deflection Separation (CDS) and Fuzzy Filter high-rate solids removal technologies, and UV high-rate disinfection. The study was conducted at the Rockland County Sewer District No.1, in Orangeburg, NY from August 1998 to January 2001.

Three different lamp systems were evaluated within the UV disinfection studies. These were:

- PCI Wedeco UV Technology (now Wedeco Ideal Horizons). This system represents newer low-pressure lamp UV systems, which takes advantage of the high power conversion efficiency of the low-pressure lamps, while getting higher UV outputs.

- Aquionics UV Technology. This system utilizes medium-pressure lamps. These are less efficient than conventional lamps but their total UV output is higher resulting in a lower number of lamps to achieve the required light intensity.

- Generic Medium-Pressure, Open-Channel System. The channel was designed to operate lamps at two different spacings: 4- and 6-inch.

The overall objective of the study was to evaluate high-rate solids removal technologies on SSO and CSO type wastewaters, and the subsequent UV disinfection of the treated wastewaters. The results are given below.

| Table 3-2. Cost Projection of Disinfection to be Implemented at the Spring Creek Facility | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | Conceptual Level Facility Disinfection Costs ($) | | | | | | | | | | | |
| | Chlorination/Dechlorination | | | Chlorine Dioxide | | | Ozone | | | UV | | |
| Peak Design Flow (cfs) | 1,250 | 2,500 | 5,000 | 1,250 | 2,500 | 5,000 | 1,250 | 2,500 | 5,000 | 1,250 | 2,500 | 5,000 |
| Capital Costs | 912,000 | 1,045,000 | 1,219,000 | 695,000 | 1,159,000 | 1,932,000 | 19,221,000 | 24560,000 | 30,539,000 | 48,052,000 | 67,272,000 | 87,774,000 |
| Annualized Capital Costs | 93,000 | 107,000 | 124,000 | 70,000 | 119,000 | 196,000 | 1,957,000 | 2,502,000 | 3,111,000 | 4,894,000 | 6,852,000 | 9,592,000 |
| Annual O&M Cost | 255,000 | 255,000 | 255,000 | 294,000 | 294,000 | 294,000 | 534,000 | 587,000 | 657,000 | 248,000 | 497,000 | 992,000 |
| Total Annualized Costs | 348,000 | 362,000 | 379,000 | 364,000 | 413,000 | 490,000 | 2,491,000 | 3,089,000 | 3,768,000 | 5,142,000 | 7,349,000 | 10,584,000 |

Notes:
1. Costs are present worth in 2000 dollars.
2. Capital costs are based upon sizing to meet peak design flow and a 4-log reduction in fecal coliform.
3. Capital costs are for installation of Spring Creek and are for process equipment only. Costs do not include additional contact tankage (if required) or support facilities.
4. Annual operating costs are based upon an assumed typical 40 CSO events/year at a volume treated of 15 million gallons per event.
5. Annualized costs are based upon a period of 20 years at an interest rate of 8%.

### *UV Disinfection Dose Requirements and Particle Size Impacts*

The dose-response analyses indicated that removal of particles greater than 50-micron in size will improve the efficiency of the UV process because filtration to such levels removes a substantial amount of occluded bacteria. Samples were blended prior to analysis to release occluded bacteria so they could be detected in analysis. Blending the unfiltered samples released fecal coliform and improved recovery of occluded bacteria. Blending samples that had been filtered at retention levels between 1 and 50 microns did not have a significant impact on coliform recovery and did not impact UV dose requirements to accomplish targeted reductions.

The UV dose requirement to accomplish 3-log reduction of fecal coliform in primary-type wastewater (i.e., wastewater of a quality equivalent to a primary-treated wastewater), pretreated to remove particles greater than 50-microns is approximately 20 $mJ/cm^2$. The results suggest that the maximum reductions that can be expected under practical dose applications up to 40 $mJ/cm^2$ are 3.5 to 4 logs. With unfiltered effluents and primary-treated wastewaters passed only through the CDS unit, the maximum reductions suggested by the dose-response analyses are approximately 2.5 to 3.0 logs (based on enumeration of blended samples).

### *CDS Process Performance*

The CDS process is capable of accomplishing approximately ten percent TSS removals with a 1200-micron screen. This increases to approximately 30 percent with a 600-micron screen. In both cases, it appears that removals were independent of the flow rate, within the range of flows tested.

The CDS unit, based on visual observations, was effective in capturing and removing debris, including paper and plastics, fibers, and preventing transport to downstream processes. In this respect, the wider aperture screens were as effective as the smaller aperture screens and are more easily maintained. The wider aperture screen tended to be self-cleaning while the smaller aperture screen required manual cleaning and tended to retain the debris on the screen surface. The CDS process can provide protection of downstream filters or other pretreatment devices by removing debris and floatables.

### *Fuzzy Filter Performance*

The filter was effective in removing larger-size suspended solids. The particle size distribution (PSD) and dose-response analyses confirmed that these removals centered on particles greater than 50 microns. The system is more effective in this application at 20-percent compression and at hydraulic loadings between 400 and 800 $Lpm/m^2$ (10 and 20 $gpm/ft^2$). At these conditions, TSS removals averaged approximately 40 %. Removals were consistently less at these hydraulic loadings for the 10 and 30 % compressions.

*UV Disinfection Performance*

The combined results generated with the three UV units indicate that a degree of disinfection with primary wastewaters can be accomplished by UV radiation. Reductions between 2.3 and 2.8 logs can be achieved at hydraulic loadings between 8 and 38 Lpm/kW of lamp input power (2 and 10 gpm/kW) based on the enumeration of blended samples. This is equivalent to approximately 3 to 3.5 logs when enumeration is conducted using standard analyses without blending samples. Doses greater than 40 $mJ/cm^2$ are required to achieve these reduction levels (U.S. EPA, 2002c).

### 3.2.6.3 Advanced Demonstration Facility (ADF) in Columbus, GA

Chlorination/dechlorination of CSO, along with several alternative technologies, were tested on over 40 wet-weather events at a full-scale ADF in Columbus, GA. The CSO testing program at ADF was a part of a multi-year watershed study sponsored by the Columbus Water Works Agency with the Wet Weather Engineering & Technology (WWETCO) firm as the principal contractor and with the involvement of the WERF and the U.S. EPA. ADF is comprised of multiple CSO technologies arranged as treatment trains: hydraulic controls, screening, vortex separation, compressed media filtration, and chemical disinfection using $Cl_2$ as sodium hypochlorite, $ClO_2$, peracetic acid, and UV disinfection. Multiple technologies were tested side-by-side and in sequential and split stream for determining performance at different loading rates and equipment settings. Performance results and relative costs are summarized below (Columbus Water Works, 2001).

*ADF CSO Technology Evaluations*

The ADF demonstration facility, with permitted capacity of 48 MGD, consists of coarse screening and flow controls, six 32-ft diameter vortex separators, a compressed media Fuzzy Filter (a 30-inch bed of 1-inch fiber balls contained between two perforated plates), a medium pressure UV system located downstream of the Fuzzy Filter (u-tube arrangement of two banks of 42 bulbs), and other auxiliary equipment. The ADF is fully automated and operates during wet-weather events when runoff exceeds interception. Manned operations include both pre-and post-event activities as well as preventive maintenance. Continuous rainfall monitoring and level instruments automatically initiate operations such as screening, underflow pumps, and disinfection equipment. Post-event activities include residuals removal from screens and grit bins, sodium bisulfite dechlorination, and equipment operation checks.

Testing of three chemical disinfection technologies, $Cl_2$ as sodium hypochlorite, $ClO_2$, and peracetic acid, was conducted in designated vortex separators for each technology. The vortex separator is designed to remove grit and concentrated solids but can be and was used for combined solids removal and chemical disinfection. Vortex has no moving parts and acts like a plug-flow reactor providing contact time greater than 70% of theoretical. There is higher usage of chemicals in a vortex than in a separate disinfection tank but the cost of additional chemicals is less than the cost of separate tankage. Sodium bisulfite dechlorination was also conducted in a vortex.

Chemical disinfectant was delivered by feed pumps according to a developed control algorithm for changeable dosing. At the ADF, the disinfectant demand for CSO was correlated with its ammonia and COD content in conjunction with the continuous flow and time measurements. Chemical disinfection efficiency also correlated with pH, temperature, and TSS. The highest disinfectant dose was given at the beginning of the event and it was diminishing as the event was progressing. A minimum contact time used was three minutes. Disinfectants listed in order of their effectiveness were $ClO_2$, sodium hypochlorite, and peracetic acid, however all were capable to accomplish a satisfactory disinfection. Chemical dosing under similar conditions requires 15 mg/L sodium hypochlorite, 16 mg/L peracetic acid, and 12 mgL of $ClO_2$.

Sodium hypochlorite was selected because $ClO_2$ requires generation onsite with the use of $Cl_2$ gas and peracetic acid is not licensed in the U.S. for wastewater disinfection. Sodium hypochlorite ($Cl_2$) dose varied from 4 to 30 mg/L with average concentrations between 8 and 9 mgL. Contact times ranged from 6 to 40 minutes at peak flow rates at events tested. The predominant contact times were between 10 and 20 minutes. Chlorine disinfectant residuals, when operating with variable feed rates, were typically around 1 mg/L. Dechlorination was designed for chlorine residuals exceeding 1 mg/L.

The compressed media filter provided a sufficient pretreatment level for UV disinfection. A double bank of medium pressure high intensity UV lamps (42 lamps per bank) reduced bacteria counts to the hundreds and thousands level (colonies per 100 mL) for flows of 10 to 20 MGD. The contact time for UV disinfection was two minutes. These results were for average conditions of TSS at 50 mg/L, 20% light transmittance and 25 degrees Centigrade water temperature. Transmissivity of treated flow was very important for UV. For example, UV disinfection of *E. coli* bacteria in filtered effluent with about 60% transmissivity was on the order of a magnitude higher (in hundreds of colonies per 100 mL) than in effluent with 40% transmissivity (in thousands of colonies per 100 mL). In contrast, the unfiltered CSO UV transmittance was as low as 20%.

A spreadsheet model was developed to evaluate combinations of intercept, storage, and flow through CSO treatment processes. The evaluation considered removal efficiencies, capital, and operational costs. The ADF findings provided performance criteria for vortex separation, $Cl_2$, $ClO_2$, and peracetic acid disinfection, and compressed media filtration followed by UV disinfection.

An optimized model of the ADF facility was developed. The optimized facility includes two 32-ft diameter vortex separators, instead of current six vortex separators, with $Cl_2$ disinfection followed by dechlorination and 2,000 cubic feet of compressed media filtration, instead of the current 1,000 cubic feet, followed by UV disinfection. The intercept capacity in this example is 10 MGD. The recommended peak flow capacity of the facility is 90 MGD.

Present worth, capital and operation and maintenance (O&M) costs were developed for various treatment trains, including the optimized facility, using 1995 construction costs and annual O&M costs based on several years of operation. Capital costs for a treatment system designed for 63% removal of TSS were estimated to be approximately $10,000 per acre of combined sewer service area; annual operating costs were estimated to be about $163 per acre. Designing the system for 80% removal of TSS increased the capital cost nearly threefold, with annual operating cost doubling. As discussed above, removal of TSS is representative of disinfection effectiveness, especially for UV (Arnett, 2003. Personal Communication).

#### 3.2.6.4 Washington, DC. Northeast Boundary Swirl Facility (NEBSF) (Disinfection Implementation)

The NEBSF, operated by the District of Columbia Water and Sewer Authority (WASA), provides treatment and disinfection for up to 400 MGD of CSO before discharging to the Anacostia River. The facility provides mechanical screening followed by three 57-ft diameter swirl concentrators. The effluent from swirl concentrators flows to a mixing chamber where sodium hypochlorite is added, usually at a dose of 5 mg/L. Sodium bisulfite is added at the end of the outfall for dechlorination, usually at a dose of 2 mg/L. Flows above 400 MGD are discharged untreated. Samples taken during CSO events at the mixing chamber and at the river outfall are analyzed for *Enterococcus* and fecal coliform. Reported counts range from less than 10 MPN/100 mL to in excess of 250,000 MPN/100 mL. The high numbers are associated with events in excess of 400 MGD and represent blending of treated and untreated CSO.

Annual operating costs for the NEBSF are estimated to about $230,000. This is based on $180,000 for labor and $50,000 for chemicals. The facility discharges on average about 100 times per year, with an average total volume of approximately 1,500 MG (Siddique, 2003. Personal Communication).

#### 3.2.6.5 Birmingham, AL. UV Disinfection at Peak Flow WWTP (Disinfection Implementation under Construction)

The Jefferson County Environmental Services Division for the City of Birmingham and about 20 neighboring communities is in the process of constructing a 350 MGD peak excess flow treatment facility. The new facility, named the Village Creek Peak Flow Wastewater Treatment Plant (PFWWTP), includes a pump station, with 360 MGD capacity, 20 surge basins with surface aeration for mixing (total capacity of 90 MG), granular, monomedia, deep bed filters with 350 MGD capacity, UV disinfection, and a 24 megawatt generating facility (primarily to power the pump station and UV). The UV system will have a total of 2,688 lamps at a peak power requirement of 7,526 kW. The total installation cost of the UV facility is estimated to be $13 million; the cost of UV equipment is about $10.7 million. Operating costs are not available (Chandler, 2003. Personal communication).

#### 3.2.6.6 Oakland County, MI. Chlorine Disinfection at Acacia Park (Disinfection Implementation)

The Office of the Oakland County Drainage Commissioner currently operates three CSO retention basins in southeastern Michigan, all of which provide treatment and disinfection of flows that exceed their storage capacity. The Acacia Park CSO Retention Treatment Basin (RTB) is a 4 MGD basin that serves a combined area of about 818 acres. Disinfection is by sodium hypochlorite. The feed system provides a dose of 10 mg/L at a CSO flow rate of 426 MGD. There is no dechlorination. The disinfection target is a fecal coliform count of less than 400 cfu/100 mL at a total residual chlorine level of 1.0 mg/L.

Annual operating costs for the Acacia Park facility are estimated to be $120,000. This includes $58,000 for labor, $24,000 for energy and utilities, $26,000 for chemicals, and $10,500 for laboratory and other services. Over the three-year demonstration period, the facility captured 60% of the flow it received; that is treated overflows represent 40% of flow into the facility. The total volume of flow into the facility was estimated at 146 MG, with 88 MG retained and returned to the sewer system and 58 MG treated and discharged. Overflows occurred on average four to five times per year, and ranged in volume from 0.13 to 17 MG (Mitchell, 2003. Personal Communication).

#### 3.2.6.7 Bremerton, WA. UV Disinfection at CSO Treatment Facility (Disinfection Implementation)

The City of Bremerton has recently constructed a CSO treatment facility that uses high-rate clarification, followed by UV disinfection, to treat flows up to 45 MGD. The facility uses a medium-pressure, high-intensity UV system that employs a total of 90 lamps. A 500-kW generator supplies power to the UV system as well as pumps, mixers, and other equipment. The clarification system uses a polyaluminum chloride coagulant. The primary reason for choosing UV over chlorination was to avoid degradation of hypochlorite between discharge events, which occur about 20 times a year. Bremerton installed a UV system at a a cost of about $600,000 to disinfect CSO discharges. The annual operation cost for the entire facility is estimated to be about $50,000 (Poppe, 2003. Personal Communication).

#### 3.2.6.8 Disinfection of Collected Stormwater and Dry Weather Urban Runoff

***New Orleans, LA - Stormwater Disinfection***

The city of New Orleans, LA evaluated a prototype disinfection facility for stormwater using sodium hypochlorite in the late 1960s and early 1970s; (Pavia and Powell, 1968) however, they did not adopt the practice permanently.

***Santa Monica Urban Runoff Recycling Facility (SMURRF)***

Santa Monica's urban runoff recycling facility (SMURRF) project, completed in December 2000, in Santa Monica, CA, treats dry weather runoff water from excessive irrigation, spills, construction sites, pool draining, car washing, the washing down of paved areas, and some wet weather runoff. SMURRF treats an average of 0.5 MGD of the above urban runoff with solids, and oil and grease removing technologies prior to UV disinfection for removal of pathogens. The treated runoff is reused for landscape irrigation and for in dual-plumbed buildings for flushing of toilets. For more information, see the Internet site at: http://Epwm.Santa-Monica.Org/Epwm/Smurrf.html.

***Moonlight Beach Urban Runoff Treatment Facility***

Moonlight Beach Urban Runoff Treatment Facility in the City of Encinitas, CA has been treating dry season urban runoff since September 2002. The facility accepts flows up to 150 gpm. The technologies used are filtration followed by UV disinfection. Coliform bacteria were reduced by over 99%. The facility does not operate during the wet season (Rasmus and Weldon, 2003).

## 3.3 Best Management Practices (BMPs) for Control of Pathogens in Urban Stormwater

### *3.3.1 Introduction*

Practices to control and manage the quality and quantity or urban runoff have become widespread. This set of practices has been labeled best management practices or BMPs. Structural BMPs are designed to function without human intervention at the time a storm event occurs (Urbonas, 1999). Wet ponds, dry ponds, constructed wetlands, filters, rooftop storage, and swales are examples of structural BMPs that can be applied to urban stormwater. Eliminating illicit cross connections between the sanitary sewage system and separate stormwater drainage system is another structural BMP. Similarly, reduction of stormwater volume that enters combined or sanitary sewer systems aids in reducing CSO and SSO volumes. These measures are distinct from the others because they involve repairing the stormwater or sewerage system, rather than erecting a structure to manage or control stormwater quality. Other practices that reduce stormwater volume known as inflow reduction techniques include disconnection of roof leaders and redirection of area and foundation drains and basement sump pumps to soils where the flow will infiltrate to the ground or groundwater. Nonstructural BMPs are generally good housekeeping practices or measures designed to institute good housekeeping for reducing or preventing pollutant deposition in a watershed, e.g., public education or regulation (Urbonas, 1999).

This section provides a detailed discussion of the application of structural and nonstructural BMPs to stormwater microbial contamination. Available data on performance of BMPs for removing microorganisms from stormwater are presented. However, quantitative results are inconclusive or unavailable for many of the BMPs.

### *3.3.2 Structural BMPs*

Wet ponds, dry ponds, constructed wetlands, filters, rooftop storage, and swales exhibit varied effectiveness for volume reduction and removal of suspended solids, metals, and nutrients. Structural BMPs have been applied to control pathogens to a lesser extent than to the other pollutants, and have produced mixed results. Often, controlling pathogens or microorganisms is a secondary goal for these BMPs, which are more routinely implemented for reducing flow volume, sediment, or nutrients. Some environmental professionals are of the opinion that these practices do not affect pathogens to a meaningful degree and, therefore, should not be implemented to obtain the goal of reducing microbial concentrations. Microorganism or pathogen removal has been reported most frequently by sand filters, wetlands, and wet detention ponds. EPA Storm Water Technology Fact Sheets for these BMP types are available on EPA's web site at http://cfpub.epa.gov/npdes/ (U.S. EPA, 2003b). The fact sheets include the following information:

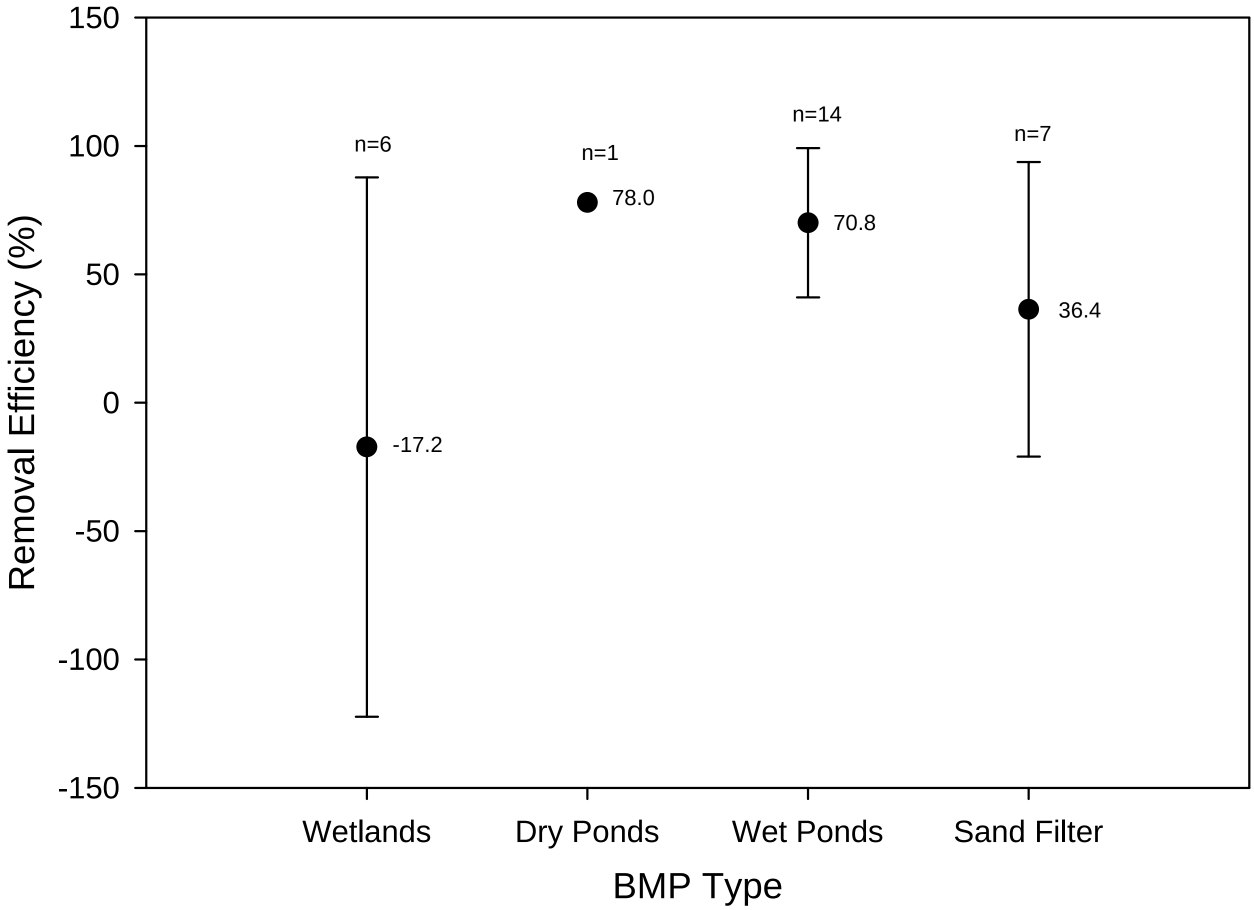
- description
- applicability
- advantages and disadvantages
- design criteria
- performance
- operation and maintenance
- costs

Limited research has been conducted on the effectiveness of structural BMPs for controlling stormwater pathogen loads to receiving waters. Much of the existing information has been compiled by Winer (2000) and by ASCE (2002) in U.S. EPA-sponsored projects. The data is compiled in database format, therefore, it is general in nature. It is included here to provide the reader with the range of BMP effectiveness and the database reference information. For more detailed information on a particular site, the reader should go to the original reference cited in the database. Reported fecal coliform removal efficiencies range from 99% at a wet pond in Ontario, Canada to –134% in a Fremont, CA wetland. These data show that while there are cases where microorganism reduction can be achieved to some extent by employing BMPs, BMPs also serve as environments where microorganisms are generated, presumably from increased wildlife populations and resuspension of bottom deposits. Table 3-3 presents performance data on the effectiveness of four types of BMPs for treating stormwater: wetlands, dry ponds, wet ponds, and sand filters (ASCE, 2002; Kurz, 1998; Winer, 2000). Figure 3-1 illustrates the variability of fecal coliform percent removal efficiencies reported. For each case study, the removal efficiencies are calculated using the average inlet and outlet fecal coliform concentrations.

| Table 3-3. Stormwater BMP Effectiveness Data. | | | | | | | |
|---|---|---|---|---|---|---|---|
| **BMP Type** | **Total Coliform (CFU/100 mL)** | | | **Fecal Coliform (CFU/100 mL)** | | | **Location and Reference** |
| | Influent | Effluent | % Removal | Influent | Effluent | % Removal | |
| Wetlands | | | | | | 78 | Lake Beardall, FL. Submerged gravel wetland. Egan *et al*., 1995, in Winer, 2000 (Study 91). |
| | | | | 2516 | 5882 | -134 | Fremont, CA. ASCE, 2002. |
| | | | | 2516 | 4581 | -82 | Fremont, CA. ASCE, 2002. |
| | 3 | 2120 | -706 | 2 | 236 | -117 | Sea Pines Plantation, SC. Surface flow, full scale, natural marsh, abundant wildlife, runoff and manure from horse trail. MacClellan, 1989, referenced in Table 17-3 of Kadlec and Knight, 1996. |
| | | | | 690 | 20 | 97 | Kingston, MA. Shallow marsh (natural or constructed not specified). Horsley, 1995, in Winer, 2000 (Study 79). |
| | | | | 1350 | 768 | 55 | Glenwood, WA. Shallow marsh (natural or constructed not specified). Koon, 1995, in Winer, 2000 (Study 80). |
| Dry Pond | | | | | | 78 | Maple Run III, TX. ASCE, 2002. |
| Sand Filter | | | | | | 37 | Joleyville, TX. City of Austin, Texas, 1990, in Winer, 2000 (Study 105). |
| | | | | | | 83 | Brodie Oaks, TX. City of Austin, Texas, 1990, in Winer, 2000 (Study 106). |
| | | | | | | 36 | Barton Creek, TX. City of Austin, Texas, 1990, in Winer, 2000 (Study 107). |
| | | | | | | 37 | Highwood, TX. City of Austin, Texas, 1990, in Winer, 2000 (Study 108). |
| | | | | 5695 | 18528 | -85 | Barton Ridge Plaza, TX. City of Austin, Texas, 1990, in Winer, 2000 (Study 109). |
| | | | | | | 81 | Barton Creek Square, TX. City of Austin, Texas, 1991, in Winer, 2000 (Study 110). |
| | | | 59.4 | | | 66 | Madeira Beach, FL. Kurz, 1998. |

| Table 3-3 continued. Stormwater BMP Effectiveness Data. | | | | | | | |
|---|---|---|---|---|---|---|---|
| **BMP Type** | **Total Coliform (CFU/100 mL)** | | | **Fecal Coliform (CFU/100 mL)** | | | **Location and Reference** |
| | Influent | Effluent | % Removal | Influent | Effluent | % Removal | |
| Wet Ponds | | | | | | 70 | Monroe Street, Wisconsin. Bannerman and Dodds, 1992, in Winer, 2000 (Study 91). |
| | | | | 83633 | 1324 | 98 | St. Elmo, TX. City of Austin, Texas, 1996, in Winer, 2000 (Study 26). |
| | | | | | | 86 | Unqua, NY. Driscoll, 1983, in Winer, 2000 (Study 34). |
| | | | | | 1779 | 90 | Heritage Park, Ontario, Canada. Liang, 1996, in Winer, 2000 (Study 43). |
| | 470 | 395.6 | 16 | | | | Jacksonville, FL. ASCE, 2002. |
| | | | | 6937 | 2516 | 64 | Fremont, CA. ASCE, 2002. |
| | | | | 17619 | 4764 | 73 | Davis, NC. FC Mass Removal Efficiency reported 48.1%. Borden *et al*., 1996, in Winer, 2000 (Study 11). |
| | | | | | | -6 | Piedmont, NC. Borden *et al*., 1996, in Winer, 2000 (Study 12). |
| | | | | | | 46 | Woodhollow, TX. City of Austin, Texas, 1991, in Winer, 2000 (Study 13) and ASCE, 2002. |
| | | | | | 783 | 64 | Harding Park, Ontario, Canada. Fellows *et al*., 1999, in Winer, 2000 (Study 16). |
| | | | | | | 56 | East Barrhaven, Ontario, Canada. Ontario Ministry of the Environment, 1991, in Winer, 2000 (Study 19). |
| | | | | | | 99 | Kennedy-Burnett, Ontario, Canada. Ontario Ministry of the Environment, 1991, in Winer, 2000 (Study 20). |
| | | | | | | 97 | Uplands, Ontario, Canada. Ontario, Canada. Ontario Ministry of the Environment, 1991, in Winer, 2000 (Study 21). |
| | | | 64 | | | 98 | Tampa, FL. Kurz, 1998. |
| Influent and effluent data provided in table when available. | | | | | | | |

Figure 3-1. Fecal Coliform % Removal Efficiency by BMP Type.




Legend: ● is mean; error bar is standard deviation. n = number of BMPs reported
ASCE, 2002; Kurz, 1998; Winer, 2000.

There are many factors affecting variability including stormwater characteristics, BMP design, and environmental factors contributing to microorganism die-off.

#### 3.3.2.1 Ponds and Wetlands

In contrast with the fact that better performance was observed in wet ponds over wetlands in the studies reviewed above, a number of research studies show that wetlands may provide advantages over ponds for indicator microorganism removals. One study found greater removal of thermotolerant coliforms, enterococci, and heterotrophic bacteria from stormwater in a wetland system (80-87%) than in a pond (–2-22%) (Davies and Bavor, 2000). The researchers attribute greater bacteria removal in the wetland to increased sedimentation aided by vegetation and increased removal of fine suspended particles (< 2 microns) with attached bacteria. Pond and wetland performance on microorganisms in sewage is an indicator of their performance on stormwater. A wastewater treatment wetland removed 97-99.9% of fecal coliform and *Enterococcus* and 70% of coliphage (Stenstroem and Carlander, 2000). The investigators attribute the bacteria concentration reductions to the wetland's ability to remove suspended particles. Viruses have been shown to accumulate in wetland biofilms resulting in their removal from the effluent (Flood and Ashbolt, 2000).

The University of Arizona sponsors a research program on constructed wetlands treatment of secondary sewage effluent at the Pima County Constructed Ecosystem Research Facility in Tuscon. Although the research examines the effect of constructed wetlands on reducing microbial pathogen and indicator concentrations in secondary sewage effluent, the results provide useful information that can be applied to stormwater. A duckweed-covered pond, a multi-species subsurface flow wetland, and a multi-species surface flow wetland reduced concentrations of *Giardia* cysts, *Cryptosporidium* oocysts, total coliform, fecal coliform, coliphage, and enteric viruses in secondary sewage effluent (Gerba *et al*., 1999; Karpiscak *et al*., 1996; Thurston *et al*., 2001). Removal of the larger microorganisms, i.e., *Giardia* and *Cryptosporidium*, was the greatest in the duckweed pond, with sedimentation thought to be the primary removal mechanism. In contrast, the greatest removal of coliforms and coliphage occurred in the subsurface flow wetland, which may be related to the large surface area available for adsorption and filtration (Gerba *et al*., 1999). When supplying potable water to a wetland at the facility, Thurston *et al*. (2001) showed that total and fecal coliform concentrations increased. The researchers attribute the greater densities found in the summer months to the flora and fauna in and around the wetland. Warm waters promote the growth of bacteria contained in the animal feces deposited in the wetland. Increased plant growth may increase root exudates, oxygen to the rhizosphere, and accumulation of organic matter, believed to increase microorganism growth (Thurston *et al*., 2001). The results of these studies are summarized in Table 3-4.

Performance of constructed wetlands treating dairy farm wastewater for use in irrigation provides another source of information related to the effectiveness of constructed wetlands on removing pathogens from stormwater runoff. Kern *et al*. (2000) conducted a seasonal effects study at a side-by-side wetland constructed at the Institute of Agricultural Engineering in Potsdam, Germany. The subsurface flow wetland with a horizontal water flow reduced fecal coliform densities by 99.3 and 95.8% in the summer and winter, respectively. The principal mechanism in eliminating fecal coliform seemed to be adsorption to soil particles followed by die-off and predation (Kern *et al*., 2000). During the summer months, vertical distribution of fecal coliform densities in the control wetland bed, which did not receive wastewater, was equivalent to the levels in the treatment bed. In the winter, fecal coliform counts were three orders of magnitude higher in the treatment bed. High counts in the control bed in the summer were attributed to the presence of warm-blooded animals.

| Table 3-4. Results of Wetlands Effectiveness Studies on Secondary Sewage Effluent at Pima County, AZ Constructed Ecosystem Research Facility. | | | | | | | |
|---|---|---|---|---|---|---|---|
| **Reference** | **Wetland Type** | **Percent Reduction** | | | | | |
| | | **TC** | **FC** | ***Giardia*** | ***Cryptosporidium*** | **Enteric Viruses** | **Coliphage** |
| Karpiscak *et al*. (1996) | Multi-Species Surface Flow | 98 | 93 | 73 | 58 | 98 | N/A |
| Gerba *et al*. (1999) | Duckweed Covered Pond | 62 | 61 | 98 | 89 | 38* | 40 |
| Gerba *et al*. (1999); Thurston *et al*. (2001) | Multi-Species Subsurface Flow | 99 | 98 | 88 | 64 | 95 | N/A |
| * from Karpiscak *et al*. (1996) reporting July - December 1994; other duckweed results reported by Gerba *et al*. (1999) for period July 1994 - December 1995 | | | | | | | |

Karpiscak *et al*. (1999) studied the effectiveness of an integrated wastewater treatment facility, consisting of solids separators, anaerobic lagoons, aerobic ponds and constructed wetlands, on dairy waste in Glendale, Arizona. In the aerobic pond, fecal coliform and *Listeria* concentrations decreased by 98.5 and 96.6%, respectively. Total coliform, however, increased by approximately 40%. Concentrations of all three organisms were decreased in the wetlands, total coliform by 79%, fecal coliform by 82.8%, and *Listeria* by 99.1%. Reductions are attributed to UV radiation, degradation of organic matter, solids settling, competition from other microorganisms, phytoremediation, and residence time.

#### 3.3.2.2 Sand Filters

Sand filters operate by trapping suspended particles or adsorbing pollutants. Sand filters can be constructed in underground trenches or in above-ground, pre-cast concrete boxes. Advantages include the lower areal requirements than ponds and the ability to install them out of public view (Kurz, 1998), both of which facilitate their use in ultra-urban environments where ponds are more difficult to site.

#### 3.3.2.3 Illicit Discharge Detection and Elimination

Improper connections to storm drainage systems convey contamination to receiving-water bodies. Sources of microbial contamination transported through this route include sanitary wastewater and septic tank effluent (Pitt *et al*., 1993). Since the 1980s, many municipalities initiated programs to identify and correct illicit connections in response to information highlighted by EPA's Nationwide Urban Runoff Program (U.S. EPA, 1983) and the 1987 Clean Water Act. The Clean Water Act requires municipal separate storm sewer system discharge permits to effectively prohibit non-stormwater discharges into storm drains. EPA has an Internet site that presents information about illicit discharges, how specific municipalities are working to address them, and methods for identifying them: http://cfpub2.epa.gov/npdes/stormwater/menuofbmps/illi_2.cfm (U.S. EPA, 2003b). Pitt *et al*. (1993) published an EPA User's Guide on investigating inappropriate pollutant entries into storm drainage systems available at http://www.epa.gov/ednnrmrl/repository/cross/cross.pdf. An update of this manual has been funded by EPA and will be published in the near future. It is a collaborative effort between Pitt and the Center for Watershed Protection. The new manual will include information on optical brightener monitoring, a quick and effective way for screening large watersheds for illicit wastewater connections.

Procedures for identifying potential illicit discharges to storm drain systems include reviewing existing drainage area maps, surveying building storm drain connections, and inspecting sewer lines (U.S. EPA, 2003b). Visible flow during dry periods is a sign of a possible cross connection that should be further investigated. Visual inspection of the insides of a sewer system can be done with television equipment. Differences between known connections shown on maps and those revealed by the television should be further investigated. Tracers are often used to investigate suspected illicit connections (Pitt *et al*., 1993). A tracer is a parameter not characteristic of the base flow; the particular tracer present is dependent on the content of the illicit discharge. Tracers include water temperature, specific conductivity, fluoride and/or hardness, ammonia and/or potassium, surfactants and/or fluorescence (including optical brighteners from laundry detergents), chlorine, color, odor, turbidity, and flotables. Tracers for microbial contamination would include sanitary wastewater parameters such as BOD or suspended solids. Tracers can also be artificial, such as a dye. Smoke testing is another investigative method for illicit connections. Zinc chloride smoke injected into the sewer lines emerges from all breaks in the sewer line, vents in connected buildings, and outfalls (U.S. EPA, 2003b).

### *3.3.3 Nonstructural BMPs*

Nonstructural BMPs include institutional and educational practices with the goal of changing behaviors so that the amount of pollutants entering the stormwater drains and receiving waters are reduced (Urbonas, 1999). These common sense measures for addressing microbial contamination include limiting public and animal access to sensitive watershed or riparian areas, public education on the role of storm drains, erosion control, vegetative buffers, street sweeping, animal waste management, and pet waste or pooper-scooper ordinances. While quantitative data on nonstructural BMP effectiveness are limited, a number of these practices have been shown to reduce receiving-water bacteria levels in rural and agricultural settings, primarily by controlling sources. They are provided here because some of the practices may apply to urban watersheds, particularly developing rural areas. Several demonstrations are described in the report prepared for EPA entitled *Section 319 Nonpoint Source National Monitoring Program – Successes and Recommendations* (Lombardo *et al.*, 2000). The types of practices reported to be successful are riparian/livestock exclusion fencing, riparian zone vegetation establishment and/or restoration, improved grazing management including stream crossings, improved handling of barnyard runoff and manure, campground educational programs on waste disposal, and upgrading septic systems. Project updates included in the 2002 update report (Lombardo *et al*., 2002) available at http://h2osparc.wq.ncsu.edu/02rept319/indexframe.html show mixed results associated with using BMPs for reducing nonpoint source microbiological contamination. Some of the relevant results are presented below.

- The following BMPs were implemented in Arizona's Oak Creek Canyon Watershed: erecting permanent barricades along a highway to significantly reduce visitor access to the watershed's state park and campground, improving restroom facilities at the park and campground, and educational outreach. While limited improvement to the water quality in Oak Creek is attributed to these BMPs, the watershed task force is investigating additional sources of fecal coliform that, if addressed, can result in further improvement.

- Reductions in fecal coliform in California's Morro Bay Watershed are attributed to measures used to restrict or eliminate cattle access to riparian pastures.

- BMPs implemented in Washington's Totten and Eld Inlets are repair of failing on-site wastewater treatment systems and implementation of farm plans on farms that potentially threaten receiving-water quality. "Freshwater fecal coliform count and loading results suggest that for Burns, Pierre, and McLane creeks, the degree of BMP installation and maintenance is inadequate, and/or that unfactored demographic change may be eroding what might otherwise be improved conditions. For Schneider and Perry creeks, where water quality improved, the ability to link the improvement to pollution-control programs is hampered by lack of a control in one case, by non-BMP land-use change in the other case, and by inadequate BMP data in both cases. If effectiveness is measured by significant lasting decreases in pollution, then the results allow the possibility of effectiveness in these two cases. In those cases where pollution decreased, it appears to be on the rise again, which suggests that nonpoint pollution-control programs need to be at least cyclical if not continuous." (Batts and Seiders, 2003).

- A system of BMPs designed to exclude livestock from critical areas of streams and riparian zones has contributed to a reduction in indicator bacteria counts from 29 to 40% in Vermont's Lake Champlain Basin Watershed. Indicator bacteria counts exhibited pronounced seasonal cycles – low in winter and high in the growing season beginning in May. Additional experiments confirmed that indicator bacteria survive in stream sediments during the warmer months and can be resuspended when the sediments are disturbed. Decreases in *E. coli* and fecal coliform occurred during all seasons in the two watersheds studied, while fecal streptococcus decreases were significant in one of the watersheds. (Meals *et al*., 2001).

- Erosion control and animal waste management practices were implemented in Alabama's Lightwood Knot Creek Watershed. Although water quality improved for a number of characteristics, fecal bacterial concentrations were not improved. Fecal coliform concentrations decreased to some extent, but not to a significant degree. Fecal streptococcus concentrations increased in the watershed. The relatively small change was attributed to a design flaw in the constructed cattle crossing that encourages cows to congregate on the crossing during dry periods. (Cook and O'Neil, 2003).

- BMPs were shown to decrease indicator bacteria concentrations in North Carolina's Long Creek Watershed. The 70% decrease in median fecal coliform levels in one part of the creek is attributed to livestock exclusion. The installation of exclusion fencing in the pasture of the area's largest dairy farm is believed to be responsible for 90% and 80% decreases in fecal coliform and fecal streptococci levels.

Aside from farm animals, indigenous wildlife, rodents, and pets can increase indicator microorganism concentrations to levels that exceed water quality standards. In Northern Virginia's Four Mile Run Watershed, microbial source tracking identified a number of species (waterfowl, raccoon, human, dog, deer, and Norway rat) as the *E. coli* sources (Simmons, Jr. *et al.*, 2000; NVRC, 2002). The TMDL developed for fecal coliform requires that loadings from waterfowl, raccoon, dog and other wildlife, as well as humans, be reduced by significant percentages (NVRC, 2002). Although nonstructural BMPs will likely be used, the TMDL document does not address how achieving the TMDL goal will be accomplished. The approach will be presented in the TMDL implementation plan to be developed.

Instituting pet waste management or pooper-scooper laws is the traditional way communities have dealt with pet waste, which can contaminate water bodies or pose a potential threat to residents through direct contact. Waye (2003) cites the success of dog parks as BMPs. These parks should be located away from water bodies, and provide fencing, public education on managing waste, and disposal bags and receptacles. Having a local community pet group take responsibility for a park and establishing the norm of picking up after one's own pets help to ensure success of these parks.

Other nonstructural BMPs include modifying storm drain inlets to impede rodent access, public education, labeling storm drain inlets, and street sweeping.

#### 3.3.3.1 Managing Waste from Resident Canada Geese

In recent years, geese populations have grown in many areas in the U.S. The problems encountered by local communities are the health and cosmetic problems associated with the fecal material generated, as well as the number of geese, and related traffic and safety concerns as these large birds cross traffic. Municipalities are instituting measures to protect public health from the impacts associated with this waste. The coastal town of Spring Lake, in New Jersey's Monmouth County, is experiencing high bacteria levels in a pond occupied by many Canada geese. During rain events, the pond overflows into the ocean, resulting in beach closures. The municipality automatically bans swimming at the nearby ocean beaches for 24 hours after it rains at least one-tenth of an inch (Bates, 2003). Restricting contact with recreational waters during wet-weather events is practiced by many municipalities as a precautionary measure because of the potential for waterborne illnesses to result in swimmers in contact with pathogens in the wet weather discharges.

Colts Neck, another Monmouth County community, recently contracted with the U.S. Department of the Interior's Fish and Wildlife Service to asphyxiate Canada geese at local ponds. The local health officer defended the action based on the nuisance and potential health hazards posed by the geese droppings in and around the ponds (Jordan, 2003). Some citizens and animal rights advocates opposed the action and proposed alternatives. Waye (2003) names the possible alternatives identified by GeesePeace (www.geesepeace.org), including egg addling, vegetative barriers around water bodies, border collie patrols, goose repellants, and "no feed" zones.

Most Canada geese populations are migratory, wintering in the U.S. and migrating north to summer breeding grounds in the Canadian Arctic. The availability of park-like open spaces with short grass adjacent to water bodies have resulted in growing numbers of locally-breeding geese in the U.S. known as resident Canadian geese. There are an estimated 3.5 million resident Canada geese in the U.S. (U.S. Fish and Wildlife Service, 2002). Resident geese are protected under the Migratory Bird Treaty Act of 1918 and the Migratory Bird Conservation Act of 1929, and cannot be legally taken during a hunting season, unless a special federal permit is obtained from the Service. The proposed draft Environmental Impact Statement (EIS) released March 4, 2002, by the U.S. Fish and Wildlife Service grants the States the authority to implement approved population control strategies, such as nest and egg destruction, and trapping and culling programs, without having to go through the permit process. Until the draft EIS is finalized, scheduled for the fall 2003, states must obtain a special permit from the Service for resident Canada geese population control strategies.

### *3.3.4 Effects of BMPs on Receiving-Water Quality*

From the available information on structural and nonstructural BMPs, it is evident that more research is needed on their effectiveness in reducing microbiological loads in stormwater runoff. Further, there should be a distinction between the effectiveness of structural and nonstructural BMPs. The highly variable effectiveness data exhibited by structural BMPs indicate that a variety of conditions affect the behavior of microorganisms and thus performance. These include BMP volume, temperature, light intensity, wetland plant type, filter design, and maintenance scope and frequency. As research in these areas progresses, BMP designs and O&M requirements can be aimed at achieving improved results. With even less quantitative information available for nonstructural BMPs, studies of their effectiveness in watersheds will provide information for health and environmental managers in other watersheds.

A concern with using BMPs to treat stormwater is that the microbial densities in the effluents may exceed water quality standards, even in BMPs considered to be performing well. For example, Davies and Bavor (2000) report a geometric mean for *Enterococcus* concentrations of $9.0 \times 10^2$/100 mL for the wetland's outflow, which is much higher than the U.S. recreational fresh water standard of 33/100 mL. In a case like this, the receiving water will need to have a high enough flow rate or volume to achieve the water quality target through dilution. Therefore, using a single BMP may not provide the level of treatment needed, in which case other options will need to be considered. These include incorporating a preliminary treatment step upstream of a structural BMP to create a treatment train or disinfecting the stormwater. Reducing runoff volume and source control are the most reliable ways to decrease microorganism loads to receiving waters from stormwater.

## 3.4 Conclusions

Managing microbial contamination in urban watersheds presents unique challenges. A primary reason for this is that some of the microorganism content in runoff and waterways occurs naturally because microorganisms are components of waste products deposited by animals residing in these watersheds. The populations of these organisms vary with animal population and are affected by environmental factors such as temperature, sunlight, and nutrient availability. Also, quantitative effectiveness results of the BMPs used to manage this diffuse source pollution are often unavailable or inconclusive. Therefore, managers relying on BMPs for allocating nonpoint source loads to achieve a TMDL goal need to be prepared to revise management plans and even allocations if monitoring data reveals that the desired results are not achieved.

Although some quantitative information on the effectiveness of structural BMPs for managing microbial contamination in stormwater is available, the amount of information is less plentiful than it is for other contaminants. Microorganism or pathogen removal has been reported most frequently for sand filters, wetlands, and wet detention ponds. However, the results are highly variable. The available wet pond fecal coliform data shows removals between 46 and 99 percent, except for one site where the removal was –5.8%. The wetlands efficiency data reviewed has an even greater range of removal efficiencies, from –134% to 97%. These results contradict some research studies with findings that show wetlands have better removal efficiencies than ponds. Research to understand the key biological, chemical, and physical processes controlling microorganism behavior in commonly used stormwater BMPs is necessary (Sullivan and Borst, 2001). This research would better define the relationships between design parameters and effectiveness and will contribute to the development of models that will predict effluent quality over a BMP's lifetime, temporal variations of effluent quality, and differences in performance due to differences in events (Sullivan and Borst, 2001). Also useful would be increased understanding of the relationships between common water quality parameters, e.g., TSS, and microbial indicators and pathogens.

Less quantitative information is available on the effectiveness of nonstructural BMPs than on the structural BMPs discussed above. EPA's Nonpoint Source National Monitoring Program generates some data that shows decreases, increases, and no change after BMP implementation. The watersheds described in the available program summaries are primarily rural in nature. Public education and pet waste management regulations and programs are other nonstructural BMPs that show promise for urban watershed management but for which quantitative performance data are needed.

Disinfection of CSO and other WWF types achieves a much greater degree of microorganism removal than BMPs. It's also been the subject of a much greater amount of research and investigation. Disinfection has been demonstrated to reduce microorganism concentrations in WWFs with high concentrations ($10^5$ to $10^7$ organisms/100 mL) by several orders of magnitude and produce effluents meeting permit discharge requirements ($10^2$ to $10^3$

organisms/100 mL). WWF disinfection generally occurs within shorter contact times than conventional wastewater disinfection, i.e., less than 30 minutes, with intense mixing to ensure disinfectant contact with the maximum number of microorganisms, and increased disinfectant dosage. Effective use of this high-rate disinfection process requires use of a treatment train, with an initial treatment of either filtration or inertial separation (e.g., sedimentation and vortex) to remove suspended solids. This is to address the phenomena of microbial aggregation and particle association/occlusion that cause decreased disinfection efficiency.

Chlorination is the only chemical disinfection technology currently used for disinfection of WWF. Although effective, this technology generates formation of chlorinated hydrocarbons, i.e., chloroform and THMs, which are suspected carcinogens. To address this concern and remove excess free $Cl_2$, the chlorination process can be augmented by dechlorination with either gaseous sulfur dioxide or sodium bisulfite solution. Other disinfection technologies investigated for CSO include UV light irradiation, $ClO_2$, and $O_3$. Of these three technologies, only UV disinfection has recently entered commercial use for WWF disinfection. Chlorine dioxide and $O_3$ have not been put to commercial use in the U.S. Removal efficiencies for the disinfection technologies discussed ($Cl_2$ , UV, $ClO_2$ , and $O_3$) achieve bacterial reductions of 99.9% to 99.99%. This is a significantly greater level of contaminant reduction than is achieved by BMPs. Although just beginning to be used for treating stormwater, disinfection of stormwater may be necessary to achieve water quality objectives in some watersheds.

A final point that should be considered is the uncertainty associated with the use of indicator microorganisms to determine pathogen reductions resulting from the use of a control technology or a BMP. Chapter 1 explores the relationships between indicators, pathogens, and waterborne illness. Although the desired reduction of an indicator microorganism density, TMDL, or water quality target is achieved by a certain technology or a management approach, there is still a possibility of public health impact due to the presence of disease-causing microorganisms, i.e., pathogens. Alternatively, the indicators may have provided a false or exaggerated indication of the presence of disease-causing pathogens and, thus, no benefit to human health was achieved through the control or management practice implemented. Watershed managers need to be aware of the limitations associated with indicators and remember the primary goal of protecting public health.

## References


Aieta, E.M., J.D. Berg, P.V. Roberts, and R C. Cooper. (1980). Comparison of Chlorine Dioxide and Chlorine in Wastewater Disinfection. *Journal WPCF* 52(4):810-822.

American Society of Civil Engineers. (2002). National Stormwater Best Management Practices (BMP) Database, http://www.bmpdatabase.org/.

Arnett, C. (2003). Personal Communication. Columbus Water Works, Columbus, GA.

Ashok, G., A. Drescher, D. Greene, P. Miller, C. Motau, and F. Stevens. (1997). Field-Testing UV Disinfection of Drinking Water. *Proceedings of the 23rd WEDC Conference: Water and Sanitation for All*, Durban, South Africa.

Bates, T.B. (2003). Spring Lake Beach Closes as a Pollution Precaution. *Asbury Park Press*, July 14, 2003. Asbury Park, NJ.

Batts, D. and K. Seiders. (2003). *Totten and Eld Inlets Clean Water Projects – Final Report*, www.ecy.wa.gov/biblio/0303010.html, Publication No. 03-03-010. Washington State Department of Ecology, Environmental Assessment Program, Olympia, WA, July 2003.

Calderon, R.L., E.W. Mood, and A.P. Dufour. (1991). Health Effects of Swimmers and Non-point Sources of Contaminated Water. *Int. J. Environ. Health* 1:21-31.

Chandler, H. (2003). Personal Communication. Environmental Services, Jefferson County. AL.

Columbus Water Works. (2001) *Wet Weather Demonstration Projects: CSO Technology Testing, Source Water Assessment and Protection, Watershed Assessment and Management*. Draft Summary Report, http://www.cwwga.org/.

Cook, M. and P. O'Neil. (2003). Water Quality Improvements Resulting from Implementation of Best Management Practices in the Lightwood Knot Creek Watershed, South Alabama, http://www5.bae.ncsu.edu/programs/extension/wqg/issues/Notes110.pdf. *NWQEP Notes*, NC State University, Water Quality Group, No. 110, August 2003.

Davies, C. and H. Bavor. (2000). The Fate of Stormwater-Associated Bacteria in Constructed Wetland and Water Pollution Control Pond Systems. *Journal of Applied Microbiology* 89(2):349-360.

Dernat, M. and M. Pouillot. (1992). Theoretical and Practical Approach to the Disinfection of Municipal Waste Water using Chlorine Dioxide. *Wat. Sci. Tech*. 25(12):145-154.

Field R. (1990). Combined Sewer Overflows: Control and Treatment, In: *Control and Treatment of Combined-Sewer Overflows*, Editor: P. E. Moffa. Van Nostrand Reinhold, New York, NY. 119-190.

Field R. (1996). Stormwater Pollution Abatement Technologies, In: *The Control and Treatment of Industrial and Municipal Stormwater*, Editor: P. E. Moffa. Van Nostrand Reinhold, New York, NY. 151-237.

Flood, J.A. and N.J. Ashbolt. (2000). Virus-Sized Particles Can be Entrapped and Concentrated One Hundred Fold Within Wetland Biofilms. *Advances in Environmental Research* 3(4): 403-411.

Gerba, C.P., J.A. Thurston, J.A. Falabi, P.M. Watt, and M.M. Karpiscak. (1999). Optimization of Artificial Wetland Design for Removal of Indicator Microorganisms and Pathogenic Protozoa. *Water Science and Technology* 40(4):363-368.

Glover, G.E. (1973). High-Rate Disinfection of Combined Sewer Overflow. In *Combined Sewer Overflow Seminar Papers*, EPA-670/2-73-077. U.S. EPA Office of Research and Development, Cincinnati, OH.

Johnson, J.D., R.G. Qualls, K.H. Aldrich, and M.P. Flynn. (1983). *UV Disinfection of Secondary Effluent: Dose Measurement and Filtration Effects.* EPA 600-9-83-009. U.S. EPA Office of Research and Development, Cincinnati, OH.

Jordan, B. (2003). Colts Neck Defends Decision to Gas 284 Canada Geese, *Asbury Park Press*, July 11, 2003. Asbury Park, NJ.

Kadlec, R.H. and R.L. Knight. (1996). *Treatment Wetlands*. Lewis Publishers, Boca Raton, FL and New York, NY.

Karpiscak, M.M., C.P. Gerba, P.M. Watt, K.E. Foster, and J.A. Falabi. (1996). Multi-Species Plant Systems for Wastewater Quality Improvements and Habitat Enhancement. *Water Science and Technology* 33(10-11):231-236.

Karpiscak, M.M., R.J. Freitas, C.P. Gerba, L.R. Sanchez, and E. Shamir. (1999). Management of Dairy Waste in the Sonoran Desert Using Constructed Wetland Technology. *Water Science and Technology* 40(3):57-65.

Katz, A., N. Narkis, F. Orchansky, E. Fiedland, and Y. Kott. (1994). Disinfection of Effluent by Combination of Equal Doses of Chlorine Dioxide and Chlorine Added Simultaneously over Varying Contact Times. *Wat. Res*. 10:2133-2138.

Katzenelson, E., B. Kletter, H. Schechter, and I. Shuval. (1976). Inactivation of Viruses and Bacteria by Ozone - Chapter 17. In *Chemistry of Water Supply, Treatment, and Distribution,* Ann Arbor Science Publishers, Ann Arbor, MI.

Kern, J., C. Idler, and G. Carlow. (2000). Removal of Fecal Coliforms and Organic Matter From Dairy Farm Wastewater in a Constructed Wetland Under Changing Climate Conditions. *J. Envir. Sci. Health* 35(8):1445-1461.

Kurz, R.C. (1998). Removal of Microbial Indicators from Stormwater Using Sand Filtration, Wet Detention, and Alum Treatment Best Management Practices. Surface Water Improvement and Management (SWIM) Program Technical Report, Southwest Florida Water Management District, Tampa, FL.

Lombardo, L.A., G.L. Grabow, J. Spooner, D.E. Line, D.L. Osmond, and G.D. Jennings. (2000). *Section 319 Nonpoint Source National Monitoring Program – Successes and Recommendations,* November 2000. North Carolina State University Water Quality Group, Raleigh, NC.

Lombardo, L.A., G.L. Grabow, D.E. Line, D.L. Osmond, and J.Spooner. (2002). *Section 319 National Monitoring Program Projects - 2002 Summary Report*. NCSU Water Quality Group, U.S. EPA - NCSU-CES Grant No. X97527401, Raleigh, NC.

Marsalek J., D. Averill, D. Muck-Mumford, R. Anhdrih, and D. Weatherbe. (1996). Field Facility for Research & Demonstration of CSO Treatment Technologies, http://www.ryerson.ca/civil/urban/techno/combined/endofpipe/10-2-9.htm.

Meals, D., R. Hopkins, S. Fiske, and R. Langdon. (2001). Lake Champlain Basin Agricultural Watersheds Section 319 National Monitoring Program Project, Final Report May 1994 – November 2000. Vermont Department of Environmental Conservation, June 2001.

Mitchell, D. (2003). Personal Communication. Hubbell, Roth, & Clark, Inc., MI.

Moffa, P. E. (1975). *Bench-scale High-rate Disinfection of Combined Sewer Overflows with Chlorine and Chlorine Dioxide* EPA-670/2-75-021. U.S. EPA Office of Research and Development, Cincinnati, OH.

Northern Virginia Regional Commission. (2002). *Fecal Coliform TMDL (Total Maximum Daily Load) Development for Four Mile Run, Virginia*, www.novaregion.org/4MileRun/TMDL/4mr_TMDL_5-31-02.pdf. Prepared for: Virginia Department of Environmental Quality and Virginia Department of Conservation and Recreation, First Submission: March 21, 2002; Revised: April 25, 2002; Accepted May 31, 2002.

Oliver, B.G. and E.G. Cosgrove. (1975). The Disinfection of Sewage Treatment Plant Effluents Using Ultraviolet Light. *Can. J. Chem. Eng.* 53:170.

O'Shea, M.L. and R. Field. (1992). Detection and Disinfection of Pathogens in Storm-Generated Flows. *Can. J. Microbiology* (38):267-276.

Pavia, E.H. and C.J. Powell. (1968). Stormwater Disinfection at New Orleans. In: *Proceedings of the 41st Annual Conference of the Water Pollution Control Federation*, Chicago, IL.

Pitt, R., M. Lalor, R. Field, D.D. Adrian, and D. Barbe. (1993). Investigation of Inappropriate Pollutant Entries into Storm Drainage Systems, EPA/600/R-92/238. U.S. EPA Office of Research and Development, Cincinnati, OH.

Poppe, J. (2003). Personal Communication. Wastewater Division Manager, City of Bremerton, WA.

Rasmus, J. and K. Weldon. (2003). Moonlight Beach Urban Runoff Treatment Facility. *Stormwater, the Journal for Surface Water Professionals*, May/June, 2003, pp.12-22.

Rex, A. (2000). How Clean is the Water? *Enterococcus* and Fecal Coliform Tell Different Stories about CSO Control. In: *Proceedings of Water Environment Federation Conference, Disinfection 2000: Disinfection of Wastes in the New Millennium*, New Orleans, LA.

Sakamoto, G. and W.L. Cairns. (1997). UV Dose-response Determination: Collimated Beam vs. Pilot Studies. In: *Proceedings of Hawaii Water Environment Association Conference*, Honolulu, HI.

Santa Monica Urban Runoff Recycling Facility (SMURRF Project). (2000). http://Epwm.Santa-Monica.Org/Epwm/Smurrf.Html.

Science Applications International Corporation (SAIC). (1998). Results of Sampling Study to Determine Effectiveness of Chlorination. Final Report for project done at Cottage Farm CSO Treatment Facility, Boston, MA.

Siddique, M. (2003). Personal Communication. CSO Control Program Manager, District of Columbia Water and Sewer Authority. Washington, DC.

Simmons, Jr., G.M., D.F. Waye, S. Herbein, S. Myers, and E. Walker. (2000). Estimating Nonpoint Fecal Coliform Sources in Northern Virginia's Four Mile Run Watershed. In T. Younos and J. Poff (ed.), Abstracts, Virginia Water Research Symposium 2000, VWRRC Special Report SR-19-2000, 248-267. Blacksburg, VA.

Stenstroem, T.A. and A. Carlander. (2000). Occurrence and Die-Off of Indicator Organisms in the Sediment in Two Constructed Wetlands. *Water Science and Technology* 44(11-12):223-230.

Stinson, M.K., R. Field, P.E. Moffa, S.L. Goldstein, K.J. Smith, and E. Delva. (1998). High-Rate Disinfection Technologies for Wet Weather Flow. In: *Advances in Urban Wet Weather Pollution Reduction. Water Environment Federation Specialty Conference*, Cleveland, OH.

Stinson, M.K., I. Wojtenko, and R. Field. (1999). High-rate disinfection techniques for combined sewer overflow. In: *Proceedings of the 26$^{th}$ Annual Water Resources Planning and Management Conference,* Tempe, AZ.

Sullivan, D. and M. Borst.(2001). Research in Urban Stormwater BMPs. *Proceedings of Third International Conference on Watershed Management*, December 2001,Taipei, Taiwan,

Thurston, J.A., C.P. Gerba, K.E. Foster, and M.M. Karpiscak. (2001). Fate of Indicator Microorganisms, Giardia and Cryptosporidium in Subsurface Flow Constructed Wetlands. *Wat. Res.* 35(6):1547-1551.

Urbonas, B. (1999). Assessment of Stormwater Best Management Practice Effectiveness. In Innovative Urban Wet-Weather Flow Management Systems, by Heaney, J.P., R. Pitt, and R. Field, EPA/600/R-99/029, http://www.epa.gov/ednnrmrl/publish/book/epa-600-r-99-029/. U.S. EPA Office of Research and Development, National Risk Management Research Laboratory, Cincinnati, OH.

U.S. Environmental Protection Agency (U.S. EPA). (1973). *Microstraining and Disinfection of Combined Sewer Overflows - Phase II*, EPA-R2-73-124 Office of Research and Development, Washington, DC.

U.S. EPA (1979a). *Combined Sewer Overflow Abatement Program Rochester, NY, Vol. II Pilot Plant Evaluations;* EPA/600/2-79-031b. EPA Office of Research and Development, Cincinnati, OH.

U.S. EPA. (1979b). *Disinfection/Treatment of Combined Sewer Overflows, Syracuse, New York.* EPA/600/2-79-134. Office of Research and Development, Cincinnati, OH.

.

U.S. EPA. (1983). *Results of the Nationwide Urban Runoff Program. Volume 1 - Final Report*. Water Planning Division, Washington, DC. December. NTIS Publication (83):185552.

U.S. EPA. (1984). *Health Effects Criteria for Fresh Recreational Waters*. A.P. Dufour, EPA-600/1-84-004.

U. S. EPA. (1986a). *Design Manual: Municipal Wastewater Disinfection*, EPA/625/1-86/021. Office of Research and Development, Cincinnati, OH.

U.S. EPA. (1986b). Ambient Water Quality Criteria for Bacteria – 1986, EPA-440/5-84-002. Washington, DC.

U.S. EPA. (1993). *Manual; Combined Sewer Overflow Control*. EPA/625/R-93/007. Washington, DC.

U.S. EPA. (1994). *Combined Sewer Overflow Control Policy*, 59 Federal Register 18688. April 19, 1994.

U.S. EPA. (2001). *Protocol for Developing Pathogen TMDLs*, EPA 841-R-00-002. Office of Water, Washington, DC.

U.S. EPA. (2002a). *Management of Combined Sewer Overflow*, EPA/600/R-02/xxx. DRAFT. Office of Research and Development, Cincinnati, OH.

U.S. EPA. (2002b). *CSO Disinfection Pilot Study: Spring Creek CSO Storage Facility Upgrade. Research Summary,* EPA/600/R-04/077. Office of Research and Development, Cincinnati, OH.

U.S. EPA. (2002c). *Continuous Deflection Separation, Fuzzy Filter and UV Treatment of SSO-Type Wastewaters: Pilot Study Results*, EPA/600/R-02/100. Office of Research and Development, Cincinnati, OH.

U.S. EPA. (2003a). Watershed Academy, Introduction to the Clean Water Act Module, http://www.epa.gov/watertrain/cwa/index.htm Office of Water, Washington, DC.

U.S. EPA. (2003b). Illicit Discharge Detection and Elimination, http://cfpub2.epa.gov/npdes/stormwater/menuofbmps/illi_2.cfm. Office of Water, Washington, DC.

U.S. Fish and Wildlife Service. (2002). Service Releases Draft EIS on Resident Canada Geese, http://southeast.fws.gov/news/2002/n02-000.html. Press Release, March 4, 2002.

Waye, D. (2003). Removing Bacteria from Runoff, From Alum to UV - Draft. NPS News-Notes, Issue 73, Sept.-Oct. 2003.

Winer, R. (2000). *National Pollutant Removal Performance Database for Stormwater Treatment Practices*, Second Edition, Center for Watershed Protection, Ellicott City, MD.

White, G.C. (1999). *Handbook of Chlorination and Alternative Disinfectants*. Wiley-Interscience Publication, John Wiley & Sons, Inc., New York, NY.

Wojtenko, I., M.K. Stinson, and R. Field. (2001a). Challenges of Combined Sewer Overflow Disinfection with Ultraviolet Light Irradiation. *Critical Reviews in Environmental Science and Technology* 31(3):223-239.

Wojtenko, I., M.K. Stinson, and R. Field. (2001b). Performance of Ozone as a Disinfectant for Combined Sewer Overflow. *Critical Reviews in Environmental Science and Technology* 31(4):295-309.